The Impact of Dredging and Spoils Disposal on Hamilton Harbour Fisheries: Implications for Rehabilitation.

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THE IMPACT OF DREDGING AND SPOILS DISPOSAL ON HAMILTON HARBOUR FISHERIES: IMPLICATIONS FOR REHABILITATION

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ABSTRACT

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The impacts of dredging and spoils disposal on Great Lakes fish communities are reviewed. Changes in physical and chemical variables may alter the productivity of plankton, aquatic macrophytes, benthos, and fish depending on the timing and location of the operation. Many of these changes have short-term negative effects and therefore may be difficult to detect. Some impacts, such as the release and remobilization of contaminants from dredged sediments, have long-term fisheries implications requiring further investigation.

The limits and constraints imposed on fisheries rehabilitation efforts in Hamilton Harbour by dredging and spoils disposal operations are also assessed. Detailed accounts of the physical and chemical characteristics of the sediments, their movements, and historic, current, and future dredging and spoils disposal practices as they affect the fisheries in the harbour are provided. Spoils disposal in shallow nearshore areas were a serious degradative influence on the fisheries historically. Current dredging and disposal practices do not appear to be detrimental to primary or secondary productivity in the system. No negative effects are anticipated from future dredging operations provided they continue along the Hamilton waterfront. However, future spoils disposal in confined facilities may seriously impinge upon fisheries rehabilitation efforts, since the shallow water littoral areas are the most suitable for fisheries restoration and the disposal of dredge spoils.

RÉSUMÉ

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L'étude porte sur les répercussions du dragage et de l'élimination des déblais sur les communautés de poissons des Grands Lacs. Des changements au niveau des variables physiques et chimiques peuvent modifier la productivité du plancton, des macrophytes aquatiques, du benthos et des poissons selon le moment et l'endroit choisis pour effectuer ces opérations. Bon nombre de ces changements ont des effets négatifs à court terme et peuvent donc être difficiles à déceler. Certaines répercussions, comme la libération et la remobilisation des contaminants contenus dans les sédiments dragués, ont des effets à long terme sur les pêches qui méritent une étude plus poussée.

Nous avons également évalué les limites et les contraintes imposées aux efforts de rétablissement des pêches dans le port de Hamilton par le dragage et l'élimination des déblais. Des comptes rendus détaillés des caractéristiques physiques et chimiques des sédiments, de leur mouvement ainsi que des méthodes antérieures, actuelles et futures de dragage et d'élimination des déblais dans la mesure où elles agissent sur les pêches dans le port sont fournis. L'élimination des déblais dans des zones peu profondes à proximité des côtes représentait une importante influence de dégradation sur les pêches antérieurement. Les méthodes actuelles de dragage et d'élimination ne semblent pas être néfastes à la productivité primaire et secondaire du système. Aucun effet négatif n'est prévu dans le cas des opérations de dragage futures pourvu qu'elles se poursuivent le long du bord de l'eau. Toutefois, à l'avenir, l'élimination des déblais dans des endroits confinés pourrait nuire gravement aux efforts de rétablissement des pêches étant donné que les zones littorales où l'eau est peu profonde sont les plus aptes au rétablissement et à l'élimination des déblais de dragage.

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1. INTRODUCTION

The United States and Canada dredge and dispose of approximately ten million m^3 of sedimentary materials from the lakes and connecting channels in the Great Lakes basin annually (IWGACPDA 1975; Sly 1977). These activities have substantial social and economic benefits, but they have also been implicated in the widespread degradation of the aquatic environment over the last two centuries (e.g. Francis et al. 1979). The potential impact on the fish community may be considerable since the majority of current dredging and disposal activity occurs in the littoral and nearshore zones which are important fish spawning, nursery, and food production areas.

Ninety percent of the dredging in the Great Lakes is undertaken to maintain, improve, or extend navigable waterways and harbours. Other common, though minor purposes include the mining of resource materials (sand, gravel, and shells) for construction aggregates, the removal of grossly contaminated sediments for pollution abatement, shoreline reclamation, and occasionally to improve hydraulic flow conditions, prepare marine foundations, and to provide clearance at intake pipes, etc. (Walker and Partners 1974). Most of the current environmental concern is focused on maintenance dredging projects because they regularly disrupt valuable shallow water habitat. These projects also present a continuing spoils disposal problem, and more importantly, they usually involve the removal of sediments contaminated with a variety of municipal, industrial, and agricultural pollutants, which may be remobilized into the aquatic environment.

Dredging and spoils disposal may affect the physical, chemical, and biological components of the ecosystem. Physical impacts include the movement of contaminated sediments from polluted areas to less contaminated sites, changes to bottom topography, resuspension of sediments, dispersal of sedimentary materials (turbidity), and the creation of floating scum and debris (IWGACPDA 1975). In addition, dissolved oxygen, nutrient and contaminant concentrations plus other water quality parameters and the productivity of plankton, benthos, aquatic macrophytes, and fish communities may be altered. The chemical and biological effects of dredging are generally longer-term than the physical impacts, and while some are aesthetically displeasing (e.g. floating scum and debris), the ultimate effects in terms of continuing and future fish production are uncertain. However, with the increasing interest in rehabilitating degraded areas in the Great Lakes, the long-term effects associated with dredging and spoils disposal and other human activities are receiving more attention from fisheries agencies.

Current policy and planning documents (e.g. Loftus et al. 1978, 1982; Francis et al. 1979, 1985; Regier et al. 1980; Bridger and Oster 1981; Harris et al. 1982) clearly indicate that future fisheries management in the Great Lakes will involve rehabilitating nearshore fish communities. Initial rehabilitation efforts have focused on severely degraded areas such as lower Green Bay (Harris et al. 1982) and environmentally sensitive areas such as Long Point Bay, Lake Erie (Regier et al. 1980; Francis et al. 1985). Complementing these activities, remedial action plans detailing a comprehensive and systemic approach to restoring "beneficial" uses in all degraded areas in the Great Lakes are expected to be completed by March 1987.

Hamilton Harbour is one of the most severely degraded areas in the Great Lakes. Historically, the harbour supported a large coldwater and an extensive warmwater fish community (Whillans 1977, 1979; Holmes and Whillans 1984). These fisheries gradually responded to a variety of stresses including exploitation, eutrophication, toxic chemicals, and dredging and spoils disposal, resulting in the loss of the coldwater fishery and a limited warmwater fishery. The most stressful perturbations were rapidly deteriorating water quality (Whillans 1979) and land reclamation along the Hamilton waterfront (Holmes and Whillans 1984).

Despite its degraded condition, there are indications that ecological improvements can be made in the harbour. Pollutant loadings from specific industrial waste streams and from municipal sewage treatment plants (STP) have been significantly reduced in the last ten years through the use of new treatment technology and a doubling of the hydraulic capacity at all four STP's discharging into Hamilton Harbour (OME 1985). As a result, water quality has improved since 1970 (Threader et al. 1986) and 43 fish species have been captured in the harbour (Holmes 1986). The discovery of a diverse and abundant fish community has generated optimism among environmental agencies that rehabilitation action can and will be implemented. It is also worth noting that the harbour is an important indicator of Canada's progress under the terms of the 1978 Great Lakes Water Quality Agreement because it is the most severely degraded area in the Great Lakes basin solely under Canadian jurisdiction (Bixby 1985).

An adequate assessment of fisheries rehabilitation options in the harbour will require some knowledge of the long-term environmental effects of dredging and spoils disposal since these activities are likely to continue in the forseeable future. Much of this information is currently available in the literature but has never been consolidated. The purpose of this report is to assess the biological impacts of dredging and spoils disposal on Great Lakes fish communities and the constraints these impacts may place on future management for an improved fishery, particularly in Hamilton Harbour. Key issues include the ultimate impact of dredging and disposal on fish production in Hamilton Harbour and the implications for fisheries rehabilitation under current and future dredging scenarios. Detailed accounts of the sediment dynamics and characteristics and the historic, current, and future dredging and spoils disposal practices as they affect the fisheries in the harbour are provided. The general impacts of dredging and spoils disposal on water quality parameters, plankton, benthos, and aquatic macrophytes as they affect fish will be documented in a literature review.

1.1 Environmental concerns of dredging and spoils disposal

The environmental effects of dredging depend on such factors as the nature and quantity of material to be moved, the equipment and techniques to be employed, and the history and characteristics of the dredging and disposal site (IWGACPDA 1975). To a certain extent the type of project (maintenance or capital works) also has a bearing. Maintenance projects normally involve the removal and disposal of recent sediment accumulations at predictable frequencies and locations. They generally occur in heavily used river estuaries and harbour areas and the sediments are often highly contaminated with heavy metals, organic compounds, PCBs, pesticides, oils, and greases (IWGACPDA 1975). Maintenance projects are more likely to have intense, short-term environmental effects than capital projects because the sediments are light and shifting, highly contaminated, and seldom recolonized by aquatic organisms after the initial dredging (IWGACPDA 1975; Bertrand et al. 1976).

Capital projects (new works) usually involve previously undisturbed sediments and are more physically disruptive to the bottom topography. These projects probably have a more intense long-term effect on the biotic community at the dredging site than maintenance projects (Walker and Partners 1974; IWGACPDA 1975; Bertrand et al. 1976) but they are less predictable in terms of frequency and location and may involve much larger sediment quantities.

Dredging equipment and practices in the Great Lakes have been developed to achieve maximum economic returns with little emphasis being given to environmental concerns (IWGACPDA 1975). The choice of equipment and methodology has traditionally been made on the basis of (1) the type of material to be dredged, (2) the physical environment of the dredge site, (3) the location and nature of the disposal site, and (4) the availability of suitable equipment (IWGACPDA 1975).

The most commonly used dredging plants fall into the two broad categories of mechanical and hydraulic dredges. Mechanical dredges dislodge and remove bottom materials through the direct application of mechanical force and deliver it to barges for transport to the disposal site. There are four types of mechanical dredges but the grab or clamshell dredge is the most frequently used because of its applicability to a wide range of maintenance and capital works projects (IWGACPDA 1975).

Hydraulic dredges use a pressure gradient across an impeller to create suction and transport a slurry of sediment and water either to a hopper or by pipeline to the disposal site (IWGACPDA 1975). The hopper dredge is the most common hydraulic dredge on the Great Lakes because it is a self contained unit which can operate remote from the disposal site with minimal interference to navigation (IWGACPDA 1975). Very few hopper dredges are found in the Canadian fleet.

Disposal of dredged materials can be accomplished by three methods: (1) open-water disposal; (2) unconfined disposal on land; and (3) confined disposal. Open-water disposal has traditionally been used wherever possible because it is the cheapest and easiest method (IWGACPDA 1975). However, because many sediments are becoming increasingly contaminated the practice of confined disposal is growing. Unconfined disposal on land has been used to enhance beaches, create wetlands, and reclaim land for parks and other development purposes (IWGACPDA 1975). It is not used frequently because the sediments must have rather specialized engineering properties and they must be virtually free of contaminants.

Fisheries agencies have a variety of concerns associated with the environmental impacts of dredging and disposal operations. Many of these impacts, including the physical destruction of habitat, redistribution and transportation of contaminated sediments from highly polluted areas to less contaminated disposal sites, increased turbidity, decreased dissolved oxygen levels, and the remobilization of nutrients and contaminants from the sediments, are common to both operations (IWGACPDA 1975). These in turn may alter the productivity of plankton, benthos, aquatic macrophytes, and fish.

Short-term impacts such as increased turbidity and depletion of dissolved oxygen are concerns because they limit fish habitat and may interfere with spawning and feeding migrations. They may also have trophic effects, altering plankton and macrophyte productivity and the species composition and abundance of the benthic community. Turbidity may modify the hatching success of fish eggs and fish growth rates which could affect community structure and function.

Habitat destruction and the movement of contaminated sediments from polluted to relatively unpolluted areas are longer-term effects. The loss of littoral and nearshore zone habitats are the most visible results of dredging and disposal operations. These areas are much more important spawning, rearing, and feeding habitats for many taxa (Table 1) than is indicated by their limited size. The movement of contaminated sediments from polluted to relatively unpolluted locations is a continuing source of contaminants to organisms at previously uncontaminated sites.

The remobilization of toxic substances and contaminants is probably the most serious problem associated with dredging and disposal activities in the Great Lakes (Bridger and Oster 1981; Seelye et al. 1982). The ability of fish and other aquatic organisms to bioaccumulate and bioconcentrate heavy metals and organic contaminants has led to concern that contaminated fish products may pose a risk to public health (Neff et al. 1978). Moreover, there is a large and expanding toxicological data base indicating that organic and inorganic contaminants can have adverse effects on fish growth, reproduction, and survival (Willford et al. 1981; Cairns et al. 1984).

Productivity effects are a fisheries concern for a variety of reasons.

Table 1. Utilization of the littoral zone by Lake Ontario fish species.

Species[a]	Spawning[a]	Larval[b] Nursery	Juvenile[b] Rearing	Adult[a] Habitat
Acipenser fulvescens	x	x	x	x
Lepisosteus osseus	x	x	x	x
Amia calva	x	x	x	x
Alosa pseudoharengus	x	x		
Dorosoma cepedianum	x	x	x	x
Oncorhynchus kisutch		x		
Salmo gairdneri		x	p	
Salmo trutta		x	p	x
Salvelinus fontinalis	x	x	x	x
Salvelinus namaycush	x	x		
Coregonus artedii	x	x		
Coregonus clupeaformis	x	x	x	
Prosopium cylindraceum	x	x	p	x
Osmerus mordax	x	x	x	
Esox americanus americanus		x	x	
Esox lucius	x	x	x	x
Esox masquinongy	x	x	x	x
Esox americanus vermiculatus	x	x	x	
Hiodon tergisus		x	x	x
Catostomus catostomus	x	x	x	x
Catostomus commersoni	x	x	x	x
Hypentelium nigricans	x	p	p	
Moxostoma anisurum		p	p	x
Moxostoma macrolepidotum	x	x	x	x
Moxostoma valenciennesi		p	p	x
Carassius auratus	x	x	x	x
Chrosomus eos				x
Couesius plumbeus		x	x	x
Cyprinus carpio	x	x	x	x
Hybognathus hankinsoni	x	x		
Hybognathus nuchalis	x			
Notemigonus crysoleucas	x	x	x	x
Notropis anogenus	x	u	u	x
Notropis atherinoides	x	x	x	x
Notropis cornutus	x	x	x	x
Notropis heterodon	x	x	x	x
Notropis heterolepis	x	x	x	x
Notropis hudsonius	x	u	u	x
Notropis stramineus		x	x	x
Notropis volucellus		x	x	x
Pimephales notatus		x	x	x

Table 1. Continued.

Species[a]	Spawning[a]	Larval[b] Nursery	Juvenile[b] Rearing	Adult[a] Habitat
Pimephales promelas	x	p	p	
Rhinichthys cataractae		x	x	x
Semotilus atromaculatus				x
Semotilus corporalis		x	x	x
Semotilus margarita				x
Ictalurus natalis		x	x	x
Ictalurus nebulosus	x	x	x	x
Ictalurus punctatus	x	x	x	x
Noturus flavus	x	p	p	x
Noturus gyrinus	x	p	p	x
Anguilla rostrata		p	p	p
Fundulus diaphanus		x	x	x
Lota lota	x	x	x	
Culaea inconstans	x	x	x	x
Gasterosteus aculeatus	x	x	x	x
Pungitius pungitius	x	x	x	
Percopsis omiscomaycus	x	x	x	x
Morone americana	x	x	x	x
Morone chrysops	x	x	x	x
Ambloplites rupestris	x	x	x	x
Lepomis gibbosus	x	x	x	x
Lepomis macrochirus	x	x	x	x
Micropterus dolomieui	x	x	x	x
Micropterus salmoides	x	x	x	x
Pomoxis annularis	x	x	x	x
Pomoxis nigromaculatus	x	x	x	x
Perca flavescens	x	x	x	x
Stizostedion canadense	x	x	x	x
Stizostedion vitreum vitreum	x	x	x	x
Etheostoma exile	x	x	x	x
Etheostoma microperca	x	x	x	x
Etheostoma nigrum	x	x	x	x
Percina caprodes	x	x	x	x
Labidesthes sicculus	x	x	p	x
Aplodinotus grunniens	x	x	x	x
Cottus bairdi		x		
Cottus cognatus	x	p	p	x
Myoxocephalus quadricornis	x			

p - Probably; u - Unknown.
a - From Christie (1982).
b - From Scott and Crossman (1973).

Macrophytes provide cover and spawning sites for fish and attract prey organisms (Jaworski and Raphael 1979). Similarly, plankton and benthic invertebrates are important dietary components of many fish species. Changes in the productivity of lower trophic levels invariably lead to alterations in the fish community (Borgmann 1985), often to the advantage of coarse, less sensitive, and less desirable fish species, as occurred in the Bay of Quinte, Lake Ontario (Minns et al. 1986).

The concerns outlined above will be discussed in the following sections. In reviewing the dredging literature, studies demonstrating effects on water quality parameters, plankton, macrophytes, benthos, and fish in the Great Lakes were highlighted. In certain instances when Great Lakes data were limited, studies from other freshwater and even marine systems form the main body of the review.

1.2 Water quality impacts

Water quality impacts at the dredging and disposal site fall largely into four categories: increased turbidity, decreased dissolved oxygen, and enhanced nutrient and contaminant recycling from the sediments. Strictly speaking, turbidity is a physical effect. However it is included in this discussion because it is frequently used as an indicator of water quality and it has a significant impact on light transmission, which may be a factor limiting primary production and macrophyte growth.

1.2.1 Turbidity

Turbidity is caused by the suspension of clay, silt, fine particles of organic and inorganic matter, plankton, and other microscopic organisms in the water column (APHA 1976). The standard method for determining turbidity is the Jackson candle turbidimeter, but since the lowest value it can measure is 25 units, more sensitive nephelometers are frequently used (APHA 1976). The candle turbidimeter is a visual method that measures the length of a light path through a water sample that just causes the flame of a standard candle to disappear against the background illumination. Data are reported in Jackson Turbidity Units (JTU). The nephelometer passes a beam of light through a water sample and measures the amount of light scattered at 90° to the original beam. This is proportional to the turbidity of the sample and is reported in Nephelometric Turbidity Units (NTU). Of the two methods, the nephelometric method has greater precision, sensitivity, and applicability over a wider range of turbidities (APHA 1976), but since there is no direct relationship between the Jackson candle turbidity and the intensity of light scattered at 90° (APHA 1976), direct comparisons of the data are not possible.

Increased turbidity due to disturbance and resuspension of the bottom sediments is the most obvious effect of dredging and disposal regardless of the equipment used or the type of project. Chemex (1973) found turbidity levels during dredging at Port Stanley, Lake Erie ranged from 65 to 82 JTU compared to normal background levels of 15 to 40 JTU. These figures are only rough estimates since they are derived from composite water column samples. A two year study of the effects of dock construction on the fishes in the Nanticoke region of Long Point Bay, Lake Erie was conducted by Chamberlain (1976, 1979). He found that mean turbidities at two shoreline stations close to dredging operations and two offshore stations varied from 13.0 to 24.6 JTU and 13.6 to 14.6 JTU respectively in 1975. Offshore turbidities dropped 3-6 JTU in 1976 due to a significant reduction in dredging activity. The validity of levels below 25 JTU is questionable because they are less than the reported lower limit of detection using the candle turbidimeter technique. Boyd and Griffiths (1985) examined the effects of dredging in the Keating Channel on water quality in Toronto Harbour. They found that turbidity varied from 19 to 22 NTU close to the dredging site during calm weather and there was little impact on turbidity in the Inner Harbour which ranged between 3-5 NTU. One observation during adverse weather showed that turbidity near the dredge increased to 32 NTU probably due to resuspended sediments washed downstream with the storm flows. Despite its small size and doubts as to the usefulness of some values, these data suggest that turbidity increases associated with dredging and disposal are site-specific in magnitude.

Many other studies have reported increased turbidity, but few provide specific data. For example, Bertrand et al. (1976) noted a significant but unspecified increase in turbidity near dredging and disposal operations in lower Green Bay, Lake Michigan. Keillor and Ragotzkie (1976) found higher levels due to dredging in Duluth-Superior harbour and the Keweenaw waterway while Magnuson et al. (1976) observed a similar effect during disposal of the spoils in Lake Superior. The effect of dredging on turbidity was site-specific in a study by Weber et al. (1982). Unspecified increases were noted in Toledo and Rochester

Harbours but not in Cleveland where currents and dilution are believed to have obscured possible changes.

Increased turbidity levels appear to be temporary, returning to, or close to background levels within a few hours or days after dredging and/or disposal operations cease (Walker and Partners 1974; IWGACPDA 1975; Sly 1976, 1977; Allen and Hardy 1980; Oster 1980; Bridger and Oster 1981; Levings 1982; Acres 1983; Hebert and Schwartz 1983). The duration of turbidity plumes is variable in time and space and subject to river runoff, wind effects, and water currents (Levings 1982). However, most dredged materials fall to the lake bed in minutes and complete settlement may occur in 2-3 h after dumping (Krishnappan 1975). Chemex (1973, 1975) confirmed this point in field studies at Bronte and Port Stanley. Consequently, elevated turbidity levels are generally localized and normally not measurable except in the immediate vicinity of the dredging and disposal sites (Oster 1980; Boyd and Griffiths 1985). This may explain the lack of useful data in the literature.

Although increased turbidity levels are generally transitory and localized there is some concern about the resuspension of fine grained colloidal and flocculated materials. These particles have the slowest settling time, for example, particles with a diameter of 0.02 mm will settle through 18 m of water in 10-30 h while particles an order of magnitude smaller frequently will not settle at all unless chemical flocculation or cationic adsorption occurs (Walker and Partners 1974). They are, therefore, subjected to greater dispersion at both the dredging and disposal sites, even in low energy currents (IWGACPDA 1975; Weber et al. 1982; Hebert and Schwartz 1983). This is important because heavy metals and organic trace contaminants have a strong affinity for sediments, particularly the smaller size fractions (Forstner and Wittman 1981). Because most contaminants are adsorbed to particulate material, dredging and disposal may effectively reload these contaminants into the aquatic environment (Oster 1980).

Other more subtle effects of turbidity include reduced heat transfer and altered water temperatures (Lee and Plumb 1974), reduced light penetration and photosynthesis, flocculation of algae, disruption of normal fish migrations and spawning beds, and the burial of bottom fauna and aquatic macrophytes (IWGACPDA 1975; Morton 1977; Sly 1977; Allen and Hardy 1980; Oster 1980; Levings 1982). The biotic impacts will be discussed in more detail in later sections, however, the available evidence suggests that elevated turbidity due to dredging and disposal is probably of little consequence on a lake-wide basis because of its limited spatial and temporal scope. Quantitatively, the contribution of sediments to the water column by dredging and disposal is minor compared to loadings from shipping, wind, and storm-induced resuspension (Sly 1977; Oster 1980), and the resulting turbidities are generally within the range of natural variability exhibited in lake ecosystems (Morton 1977).

1.2.2 Dissolved oxygen

Reductions in dissolved oxygen levels in the water column may occur at the dredging and disposal sites. Dredging activities generally expose deep anoxic sediments to an aerobic environment in which oxidation occurs, increasing the oxygen demand (Oster 1980). This oxygen demand may be further augmented by bacteria attacking organically enriched resuspended sediments (Morton 1977). Most of the evidence in the literature indicates that depletion at the dredge site is a short-term effect (Walker and Partners 1974; IWGACPDA 1975; Morton 1977; Allen and Hardy 1980; Oster 1980; Acres 1983; Hebert and Schwartz 1983), but the supporting data are not shown.

The evidence with respect to reductions at open-lake disposal sites is somewhat ambiguous. Bertrand et al. (1976) noted that dissolved oxygen declined an unspecified amount at a site in lower Green Bay. However, Chemex (1975) did not find a marked decrease associated with disposal activities at Port Stanley or Bronte Harbour. Different sediment organic loads may in part account for these observations. Levings (1982) reviewed the available literature and concluded that the available dissolved oxygen data provided little evidence of reductions at disposal sites.

As dissolved oxygen levels in the overlying water column decline, the E_h (a measure of redox potential) also drops (Wetzel 1975; Jennett et al. 1980). This encourages the mobilization of heavy metals from the sediments (Jennett et al. 1980; Hebert and Schwartz 1983). But only those chemical release processes and reactions with rates measurable in the order of minutes or a few hours appear to have significance (Sly 1977) because dissolved oxygen depletion is a short-lived phenomenon.

In most cases, if the water is normally oxygenated, then lowered dissolved oxygen levels will not be of sufficient magnitude to pose any problem to fish or other aquatic organisms (Acres 1983). However, thermal stratification prevents the reoxygenation of the hypolimnion except for some diffusion because the stratified water layers are exceedingly resistant to mixing (Wetzel 1975). Dredging activities that occur in the hypolimnion after stratification, particularly open-lake disposal, could add significantly to the oxygen demand from other oxidative processes in the hypolimnion (Sly 1977). This may hasten hypolimnetic anoxia and limit coldwater fish production in these areas. To date no research on this aspect has been reported in the literature.

Dredging and disposal generally cause a temporary reduction in dissolved oxygen concentrations in the water column which may encourage the mobilization of heavy metals from the sediments. Although its significance is not certain, the available evidence suggests that the contribution of heavy metals in this manner is of minor importance compared to loadings from industrial and municipal effluents. Increased oxygen demand in the hypolimnion, particularly as a result of open-water disposal, is potentially the most serious impact. The consequences of this are not clear, however, reduced oxygen levels will reduce E_h which in turn affects chemical speciation and may enhance the bioaccumulation of contaminants from the sediments. It may also hasten the development of anoxic conditions, limiting the availability of habitat, and potentially fish production, especially coldwater fish production.

1.2.3 Nutrients

Many harbour sediments contain high levels of nutrients (phosphorus (P) and nitrogen (N)) as a result of high loadings from municipal and agricultural sources. Although there is little precise information, concern has been expressed that disturbance and resuspension of the sediments by dredging and open-lake disposal may release pore waters thereby adding nutrients to the water column (Walker and Partners 1974; IWGACPDA 1975; Sly 1977; Burks and Engler 1978; Allen and Hardy 1980; Oster 1980; Levings 1982; Weber et al. 1982; Acres 1983). Chemex (1973) found significant increases in total (TP) and soluble reactive P (SRP) in the pore water and water column at Port Stanley and Bronte Harbours due to dredging activities (Table 2). The largest increases in SRP and TP occurred in the water column at Port Stanley and Bronte Harbour respectively. Soluble reactive phosphorus increased from 6 ug/L before to 29 ug/L after dredging and TP increased from 10 ug/L to 73 ug/L. In contrast, Griffiths (1983) found no evidence that dredging activities in the Keating Channel caused an elevation in P concentrations. The highest mean levels of 423 ug TP/L and 140 ug SRP/L were recorded in the Don River upstream of the dredging site. Values at the dredge site were 184 ug TP/L and 21 ug SRP/L while background levels in the Inner Harbour varied from 31 to 37 ug TP/L and 3 to 6 ug SRP/L. Similar total phosphorus data in Toronto Harbour were reported by Boyd and Griffiths (1985). The obvious influence of the Don River may have prevented the isolation of specific dredging effects in both studies.

Nutrient releases at the dredging site, particularly P, are not expected to persist. The cycling of P between the oxygenated surface waters and reduced sediments is closely linked to that of iron (Fe) and manganese (Mn) (Wetzel 1975; Burks and Engler 1978; Weber et al. 1982). Concurrent with P release, dredging also causes the mobilization of reduced, divalent Fe and Mn from the sediments and interstitial waters (see data in Chemex 1973) and their behaviour follows well defined pH/redox potential controls. The mixing of oxygenated surface water with sediments and interstitial waters containing divalent Fe and

Table 2. Mean total and soluble reactive phosphorus concentrations (ug/L) before and after dredging at Port Stanley and Bronte Harbours in 1973.

Location	TP		SRP	
	Before	After	Before	After
Bronte				
Pore Water[a]	54.8	169.4	18.1	56.0
Water Column[b]	9.5	73.4	3.3	10.0
Port Stanley				
Pore Water	57.4	161.2	18.1	61.5
Water Column	22.0	73.8	6.2	29.2

Note: Data taken from Chemex (1973).

a - Pooled sample of all segments of the core.

b - Pooled samples from several stations obtained at 5 depths in the water column.

Mn results in the rapid oxidation of Fe and Mn to relatively insoluble hydroxide forms. These particulates quickly scavenge much of the P from the water column inducing precipitation into the sediments (Wetzel 1975). This accounts for the rapidity with which elevated Fe, Mn, and P levels returned to "normal" in the water column and increases of Fe, Mn, and P in the surface sediments at Port Stanley (Chemex 1973; Sly 1977).

Chemex (1975) used two lake column simulators (one mixed and one thermally stratified) to study the release of nutrients and algal growth after dumping dredge spoils through the water column. Sediments from Port Stanley were added to the columns and measurements of orthophosphate (PO_4^{-3}) taken. There was no apparent net release of PO_4^{-3} in the mixed column compared to concentrations before sediment addition. The stratified column showed indications of PO_4^{-3} release during the first four days after the sediments were added but concentrations declined as an algal bloom developed. Subsequent PO_4^{-3} measurements showed that stratification was a key element in determining the prolonged release of PO_4^{-3} and sustaining algal growth. This seems to be confirmed by Weber et al. (1982) who showed that higher nutrient releases occur under anoxic compared to oxygenated conditions.

These data suggest that short-term nutrient releases can occur during dredging, and it is evident that on a local basis there is potential for an impact upon plankton (Kaplan et al. 1974; Sly 1984). Oster (1980) and Levings (1982) suggested that nutrient releases may lead to algal blooms and eutrophication in confined areas of lakes. Acres (1983) suggested that nutrients released by dredging in the Keating Channel could temporarily change the trophic status of the water and the fish community in favour of warmwater species.

An appendix in IWGACPDA (1975) provides evidence of the persistent effects of nutrient release at an open-lake disposal site in Lake Superior near Thunder Bay. The dumpsite is in about 30 m of water (reworking of dumped materials due to wave turbulence is considered negligible) and had not been used for 12-18 months prior to an acoustic survey in May, 1975. This survey showed that the lake bed was generally flat and the water column overlying it essentially clear. A second survey within 12 months of dumping found numerous acoustic reflectors, typical of dense fish populations in the water column above the dumpsite and evidence of recent dumping. The implication is that some form of food source from the recently dumped materials has either directly or indirectly attracted a large fish population. Adjacent areas containing older dredge spoils no longer appear to be releasing large quantities of nutrients and do not attract fish.

The effect of dredging and disposal on nitrogen (N), the other major nutrient, has not been extensively investigated. Nitrogen generally accumulates in the sediments in the reduced form, ammonia (NH_3), especially when the hypolimnion becomes anoxic (Wetzel 1975; Burks and Engler 1978). Under anaerobic conditions NH_3-N is extremely stable and can accumulate to relatively high levels (Burks and Engler 1978). However, because it is water soluble (Wetzel 1975), NH_3 is readily released into the water column during dredging and disposal and may temporarily reach levels toxic to most aquatic organisms (Burks and Engler 1978). Mortalities are unlikely in most water bodies because NH_3 is normally rapidly dissipated and diluted by large quantities of water. Ammonia may be oxidized to NO_2 and NO_3 but since this process is quite slow, dilution is probably responsible for the quick return to ambient conditions (Wright 1978). The significance of dredging and disposal as a source of nitrogen has not been assessed.

Nutrient releases at the dredging site are not expected to have long-term effects because P is rapidly scavenged by Fe and Mn hydroxides and NH_3 is diluted. Short-term impacts could include algal blooms and enhanced eutrophication especially in confined areas, but the significance of these is unknown. Long-term effects have been noted in association with disposal, particularly open-water dumping, however, they appear to be localized in nature.

1.2.4 Contaminant remobilization and release

The majority of heavy metals entering aquatic freshwater environments are associated with insoluble metal oxides or sulfides, organic matter, carbonates, clays, and other particulate material (Neff et al. 1978). Similarly, most pesticides and PCBs are usually tightly bound to the sediments (Brannon 1978) where they have a high affinity for clays, oils, and natural organics (Krantzberg and Bailey 1983). Consequently contaminant concentrations in the sediments are often several orders of magnitude greater than those in the overlying water.

The contaminant holding capacity of most materials is correlated with grain size. The smaller size fractions tend to demonstrate proportionately higher contaminant concentrations than larger particles (Krantzberg and Bailey

1983; Weber et al. 1982). This is important because these smaller sized particles are readily resuspended, have the slowest settling time, and are therefore subject to greater dispersion and contact with the water column as a result of dredging and disposal activities (Weber et al. 1982). In addition, the <70 um fraction is the particle size class most frequently ingested by the majority of deposit feeding benthos (Krantzberg and Bailey 1983). Although contaminant mobilization from the sediments is probably small compared to the total accumulations in the sediments, the environmental impacts may be considerable given the associated bioaccumulation, biomagnification, and toxicity characteristics of many contaminants (Jennett et al. 1980).

The potential release and/or remobilization of contaminants from the sediments is dependent on their physicochemical form. The mechanisms responsible for metals accumulation on or in the sediments include the following (Gambrell et al. 1976; Patrick et al. 1977; Neff et al. 1978; Krantzberg and Bailey 1983):

1) absorptive binding and cation exchange;
2) organic and inorganic complexation;
3) precipitation with carbonates;
4) precipitation with Fe and Mn hydroxides;
5) the formation of insoluble sulfides; and
6) incorporation within the crystal lattice structure of minerals.

These mechanisms and trace metal availability are influenced to a considerable degree by pH and redox potential controls. Metals bound into the structure of minerals are resistant to mild chemical attack and are essentially immobile (Patrick et al. 1977).

Although most sediments can be characterized as a "sink" for many contaminants, particularly trace metals, there is two-way exchange across the sediment-water interface. The medium of this movement is the interstitial or pore waters (Forstner and Wittmann 1981). The magnitude of exchange generally depends on the characteristics of the sediments and the overlying water and is a function of several processes including complexation, ion exchange, adsorption and desorption, mineralization and immobilization, precipitation, and other reactions (Patrick et al. 1977). Forstner and Wittmann (1981) summarized four types of chemical changes responsible for the remobilization of metals from the sediments:

1) elevated salt concentrations [hardness], whereby the alkali and alkaline earth cations can compete with the metal ions sorbed onto solid particles;

2) changes in the redox conditions [E_h], usually in conjunction with a decrease in the oxygen potential. Iron and manganese hydroxides are partly or completely dissolved, part of the incorporated or sorbed heavy metal load being released;

3) lowering of pH which leads to a dissolution of carbonates and hydroxides, as well as to increased desorption of metal cations due to competition with H^+ ions; and

4) increased use of natural and synthetic complexing agents [EDTA, NTA], which can often form soluble metal complexes sometimes of high stability with heavy metals that are otherwise adsorbed to solid particles.

Contaminant release from the interstitial waters may be aided by physical and biological processes as well (Sly 1977; Schubel et al. 1979; Jennett et al. 1980; Hebert and Schwartz 1983; Krantzberg and Bailey 1983). Physical factors include wind-wave effects in shallow high energy environments, water currents, bioturbation (sediment mixing by benthic fauna), and microbial gas production (bubbles migrate through sediments adsorbing contaminants) while biologically the methylation of metals such as Hg, As, Pb, and Sn have had significant impacts on Canadian fisheries (Wong et al. 1975, 1977; Maguire et al. 1982; Rudd et al. 1983; Maguire 1984).

Disposal of dredge spoils in water with a higher hardness than the dredge site may result in the release of metals from cation exchange sites, or the coprecipitation of metals by carbonates (Krantzberg and Bailey 1983). Laboratory work by Weber et al. (1982) tends to support the second explanation. They investigated the effects of Ca^{+2} at concentrations of 5, 50, and 120 mg/L on the release of PO_4, DOC, TKN, and several metals from the sediment. Iron, PO_4, Pb, Cr, and As release varied inversely with Ca^{+2} levels while the data for Cd, Zn, Mn, and TKN were inconclusive. Of all the constituents studied, only Mg release increased with increasing Ca^{+2} concentration. Since many of these constituents are likely sorbed to soluble colloids of Fe, OH, PO_4, and Ca, the addition of extra Ca^{+2} would be expected to cause charge neutralization and destabilization resulting in the coagulation and precipitation of these colloids. This may explain the tendency of Ca^{+2} to suppress the release of many constituents. Weber et al. (1982) also suggested that competitive cation exchange between Ca^{+2} and Mg^{+2} is the most likely explanation for the observed behaviour of Mg.

The effects of hardness during dredging and spoils disposal is dependent to a certain extent on the degree of sediment dispersion resulting from these operations. Dispersion in open-water disposal would tend to prevent coagulation of colloidal particles while spoils in confined areas may be more subject to aggregation (Weber et al. 1982). Weber et al. suggest that the addition of Ca to promote aggregation and limit metals released from confined disposal facilities (CDF) in water bodies with low hardness such as Lake Superior may be a useful technique.

The release of many metals is largely dependent on the redox environment of the sediment-water system. Khalid et al. (1977) reviewed the literature on trace metal speciation as affected by redox potential. Modifications of the redox potential (E_h) may alter metal speciation through direct changes in the oxidation state of the ion and the availability of ligand complexing molecules (Krantzberg and Bailey 1983). Redox is normally associated with dissolved oxygen concentrations; strongly oxidizing environments typically have high oxygen concentrations while a reducing atmosphere has much lower oxygen levels (Wetzel 1975). It is important to note, however, that "redox and anoxia are separable phenomena; while anoxic environments will always have a reducing atmosphere, a reducing atmosphere is not necessarily an indication of anoxia (Hebert and Schwartz 1983).

The sediments of frequently dredged areas such as harbours usually have a reductive atmosphere (Weber et al. 1982; Hebert and Schwartz 1983). The reduction of organic and inorganic compounds generates large quantities of reduced compounds, particularly sulfides (Krantzberg and Bailey 1983). Many potentially active heavy metals including Cd, Cr, Cu, Pb, and Zn are stabilized and immobilized in the sediments as insoluble sulfide compounds (Burks and Engler 1978). Lower redox levels promote the reduction of hydrous Fe and Mn oxides and the production of divalent, more soluble ions (Krantzberg and Bailey 1983). Both Fe and Mn are sensitive to changes in E_h and, along with Hg are the only trace metals in the sediment-water system that change valence as a result of changing pH and redox potential (Gambrell et al. 1976). The remobilization of Fe, Mn, and other metals coprecipitated with Fe and Mn oxides in reduced sediments followed by upward diffusion may account for the accumulation of metals in the interstitial waters and at the oxidizing-reducing interface (Munawar et al. 1984) as observed at Port Stanley by Chemex (1973).

Considerable quantities of metals particularly divalent Fe and Mn, may be present in the reduced sediments and interstitial waters. When large quantities of oxygenated surface waters are mixed with sediments and interstitial waters containing these metals, as occurs during dredging and open-water disposal, rapid oxidation and release of many trace metals may occur (Gambrell et al. 1976; Burks and Engler 1978; Hebert and Schwartz 1983). Most are probably quickly scavenged and coprecipitated with Fe and Mn hydroxides produced by the concurrent oxidation of divalent Fe and Mn (Burks and Engler 1978). Some may also be precipitated by the formation of other low solubility oxides. If the disposal environment is anoxic, as may occur in confined facilities, the low E_h may result in some dissolution of Fe and Mn hydroxides and the release of coprecipitated metals (Krantzberg and Bailey 1983). Unfortunately few studies have monitored redox potentials during dredging and disposal operations, consequently supporting data are not available.

Metal releases are also significantly influenced by pH through its effect on ion exchange, sorption, and complexation reactions (Krantzberg and Bailey 1983). Low pH (high H^+ concentrations) leads to competition for absorption sites and increased desorption of electrostatic associations with sediment particles (Munawar et al. 1984). Decreased pH may also result in the dissolution of carbonates and hydroxides with trace elements (Krantzberg and Bailey 1983).

The effect of pH is metal specific. Payne and Pickering (1975) found almost total removal of Cu (II) ions from a solution at pH >6 while at pH <3.5 no Cu ion uptake occurred. The absence of uptake at low pH was attributed to protons successfully competing for sorption sites. Farrah and Pickering (1977) noted that clay suspensions removed all Cu, Cd, Pb, and Zn ions from solution at pH 6-8 while at pH <4 there was little or no removal. Similar observations were noted by Frost and Griffin (1977). Schindler et al. (1980) studied the effects of acidification within enclosures on the mobilization of heavy metals and radionuclides from the sediments of a lake in the Experimental Lakes Area. The release of Al, Mn, Zn, and Fe occurred at pH 5 and 6. Radionuclide data showed that as pH decreased from 6.7-6.8 to pH 5.1 several metals including Fe, Co, Mn, and Zn became more soluble. Conversely, the solubility of V and Hg decreased. Increasing acidification slowed losses of Mn and Zn to the sediments while losses of Ba, Se, Cs, and V increased. A series of laboratory studies by Weber et al. (1982) compared the release of Mg, Cd, Pb, Cu, Zn, Fe, Mn, Cr, and As from sediment at pH 6.0 and 8.0. Their data show that with the exception of Cu, for which the data were inconclusive, metal releases were between 25 and 400 percent greater at pH 6.0 compared to pH 8.0.

The pH of surface waters and dredged spoils may be altered by the mixing of materials of different pH during dredging or disposal operations (Hebert and Schwartz 1983). If materials from an alkaline environment are disposed in an acidic environment (e.g. pH 5.0 or lower) metal flux to the water column should be expected (Krantzberg and Bailey 1983). Chemex (1973) found that the pH in the top 10 cm of sediments increased from pH 6.6-6.8 to pH 7.0 after dredging at Bronte Harbour while at Port Stanley a slight decrease from pH 6.8-7.0 to pH 6.8 may have occurred.

Changes in the pH of surface waters during dredging and disposal operations appear to be site-specific. Chemex (1973) noted an increase from pH 7.7 to pH 8.1 as a result of dredging in Port Stanley and Bronte Harbours. Field studies by Weber et al. (1982) found both slight increases and decreases in pH (about 0.1-0.2 units) associated with dredging and dumping operations in Cleveland, Rochester, and Toledo Harbours. The release of metals from the sediments was insignificant, however, they may have been obscured by river flows in all three harbours.

The current evidence in the dredging literature suggests that temporary changes in sediment and surface water pHs will occur as a result of dredging and open-water disposal. The magnitude and direction of change appears to be site-specific. It is not expected, given the slight changes observed elsewhere, that significant quantities of metals would be remobilized from the sediments solely due to pH changes. However, disposal into confined facilities, which potentially can develop an acidic environment (e.g. pH<5.0), may result in significant releases and subsequent escapement into the surrounding, relatively less contaminated waters.

The speciation of metals is not entirely dependent on pH or redox conditions alone. Rather it is the relationship between these factors plus water hardness and their interactions with the metals being considered that will determine the full range of chemical states possible and subsequent release and availability (Hebert and Schwartz 1983). In most cases, conditions found in the aqueous and sedimentary environments of the dredging and disposal site will limit metal speciation so that not all possible states will occur (Hebert and Schwartz 1983).

Very few studies have monitored pH, redox potential, and water hardness in relation to the release and availability of metals during dredging and disposal activities. Gambrell et al. (1976) concluded that dredging and open-water disposal would likely have little effect on pH-redox potential controls of the bulk solid materials. Although some changes may occur in the interstitial water and solids mixed with oxygenated surface waters for extended periods, the effects on trace metal solubility and availability were not considered significant. However, further studies monitoring pH, redox, and water hardness are necessary before a complete assessment of the extent of metal release and availability from sediments as a result of dredging activities can be made (Hebert and Schwartz 1983).

Natural complexing agents such as humic and amino acids and synthetics such as ethylenediamine tetraacetic acid (EDTA), nitrilotriacetic acid (NTA), and sodium tripolyphosphate (TPP) compete with sediments for dissolved metals (Munawar et al. 1984). Dredging and disposal of sediments containing these substances may result in a portion of the metal load persisting in solution where biotic uptake is more likely (Krantzberg and Bailey 1983). The significance of dredging operations on metal releases by this mechanism has not been assessed.

Jennett et al. (1980) defined the "worst case" scenario for contaminant mobilization in freshwater sedimentary systems. Their description of these conditions included the following characteristics:

1) high metal concentrations in the surface sediments;
2) extremely flocculent surface mud (e.g. predominately small size classes <5 um);
3) low redox potential (highly reducing environment);
4) high degree of mixing at the sediment-water interface;
5) elevated levels of complexing agents in the overlying water column; and
6) low pH (high H^+ concentration) in the overlying waters.

PCBs, chlorinated hydrocarbon pesticides, and other persistent organic compounds tend to accumulate in the sediments to concentrations considerably greater than in the overlying water column. As with heavy metals, concern has been raised that the resuspension of the sediments by dredging and disposal operations may lead to the mobilization of these substances into the water column where bioaccumulation and biomagnification may occur. Since these compounds are normally tightly bound to the smaller size fractions in the sediments (Brannon 1978) such concerns may be justified.

The sediment-water interactions of PCBs, pesticides, and other organics are generally specific to each class of compound and have been reviewed in detail by Munawar et al. (1984). The partitioning of these substances within the aquatic environment is governed by the water solubility and the ionization and vapourization potential of each compound plus the presence of complexing agents (Krantzberg and Bailey 1983). Other factors such as pH, water temperature, salinity, and organic pollution are only of minor importance in the uptake and release of pesticides from the sediments (Huang 1974). The factors influencing the sediment-water interactions of a variety of persistent organic compounds have been reviewed in more detail by Munawar et al. (1984).

Once organochlorine compounds (e.g. DDT and PCB) become associated with the fine sediment particles their fate is essentially dependent on the transport of these particles (Sodergren 1984). Owing to the strength of this association and properties of these compounds, direct release into the water column is not expected (Sodergren 1984). Fulk et al. (1975) conducted a laboratory study of the transfer of organochlorine pesticides from the sediments to the water column during dredging and disposal using sediments and water from three freshwater and two saltwater locations. Analyzing for aldrin, dieldrin, endrin, lindane, 2,4-D, DDT (and homologs), toxaphene, and PCBs, desorption and settling tests showed that the transfer of organochlorine compounds from the sediments to the water column was insignificant, especially at the sediment:water ratios of 1:4-1:15 used to simulate dredging and disposal conditions. Although the solids that remained suspended had high levels of organics sorbed to them, concentrations were expected to return to near background levels after 24 h of settling. Similarly, Sodergren (1984) found that most of the DDT and PCB residues in Lake Trummen, Sweden were associated with the sediment particles and only rarely could they be detected in the water column or in suspended matter before, during or after dredging.

Although laboratory results may not completely simulate field conditions and field evidence is limited, the available data appear to show that dredging and open-water disposal will not result in increased organic contaminant concentrations in the aqueous phase. However, increased contaminant levels in the water column will result from the resuspension and slow settling times of contaminated sediments. The smaller particles contain proportionately greater contaminant loads and are more frequently ingested by zooplankton and benthic organisms than are larger particles. Consequently, bioaccumulation and biomagnification of these compounds will still occur.

Lee (1975) and Brannon (1978) evaluated the environmental effects of dredging and disposal activities and concluded that they posed no hazard in terms of contaminant mobilization into the water column from the dredged material. However, they did not provide any supporting field or laboratory evidence to corroborate their conclusions. Current evidence partially supports their claim with respect to many organic compounds, but there are data demonstrating that heavy metals may be remobilized, although the magnitude of these releases has not yet been adequately assessed. Sly (1976, 1977) argues that contaminant releases by biological mechanisms, wind/wave action, storm seiches, and possibly shipping are by far the most important sources of internal loading in the Great Lakes system. This may be true on a lake-wide basis, but frequent dredging and disposal could be a significant source of pollutants in confined areas such as sheltered bays, harbours, and river mouths.

1.3 Productivity effects

Information specific to the impacts of dredging and disposal on biological communities (plankton, benthos, macrophytes, and fish) in the Great Lakes is limited (IWGACPDA 1975). The majority of available data are from studies that were not designed to address the ecological impacts of dredging activities (Levings 1982). Consequently, little progess has been made towards understanding and developing the predictive capability necessary to make ecologically rational decisions protecting renewable resources. In the absence of definitive data many impacts are more imagined than real (Allen and Hardy 1980). The following sections provide an outline of the likely ecological impacts of dredging and open-water disposal.

1.3.1 Plankton

Few field studies have quantitatively investigated the response of plankton communities (both phytoplankton and zooplankton) to dredging operations. Studies of this kind are exceedingly difficult to conduct because most plankton have a patchy, non-random distribution and are constantly moving in passive transport by water currents (Windom 1976; Sullivan and Hancock 1977). The resolution of current censusing techniques is such that 30% of a population could perish without being detected (Sullivan and Hancock 1977). For these reasons studies that demonstrate an impact on the plankton community, particularly negative impacts, should be critically evaluated and interpreted.

Several Great Lakes studies have suggested that increased turbidity may limit light penetration through the water column, reducing the depth of the euphotic zone and ultimately phytoplankton photosynthesis and productivity (Gannon and Beeton 1969; Walker and Partners 1974; IWGACPDA 1975; Bertrand et al. 1976; Magnuson et al. 1976; Oster 1980). Similar suggestions were made in reviews of other freshwater systems by Morton (1977), Allen and Hardy (1980), and Levings (1982) and in work on marine systems by the U.S. Fish and Wildlife Service (1970), Kaplan et al. (1974), and Windom (1976). Despite this consensus, it is doubtful whether turbidity (associated with dredging) would significantly reduce productivity. Most turbidity increases associated with dredging and disposal operations are temporary and generally well within the range of natural variability exhibited by lake ecosystems (Morton 1977). Potential changes would not be detected because the time scales involved are too short for current analytical techniques. In addition, certain marine phytoplankton (and presumably some freshwater species) are capable of adopting a "resting stage" under low light conditions, which reduces the effects of light limitation due to turbidity (Stockner and Antia 1976).

Conversely, other studies have argued that nutrients released from disturbed sediments may stimulate phytoplankton productivity (Bertrand et al. 1976; Windom 1976; Wright 1978), which could lead to algal blooms, especially in confined water bodies (Oster 1980; Levings 1982; Acres 1983). Chemex (1975) observed high chlorophyll a levels following dredging and disposal at Port Stanley and an algal bloom in a lake column simulator following exposure to sediments from Port Stanley. These were not long-term effects and have not been documented elsewhere probably because nutrient releases from sediments are short-lived and any concurrent changes in productivity would likely be below the limits of reliable detection.

Increased turbidity may also have a direct effect on zooplankton communities. Most zooplankton are filter-feeders. Resuspended sediments may interfere with the effectiveness of feeding appendages and increase the ingestion of non-nutritive inorganic particles (Kaplan et al. 1974; Sullivan and Hancock 1977). Flocculating sediments may adhere to eggs and adults causing cellular damage and abnormal settling to the bottom (Bertrand et al. 1976; Sullivan and Hancock 1977). Contaminants transported by resuspended particles may be ingested by zooplankton, concentrated in the food chain, and reintroduced into the system.

Turbidity effects on zooplankton production have not been extensively studied. Levings (1982) assumed that production losses were proportional to the biomass removed by mortality, but he did not provide any supporting data. Mortality, in turn, is proportional to the rate of movement with the water mass past the dredge and dumpsites, assuming entrainment occurs in the immediate vicinity of these sites. If sensitive or important life stages are present, then more than simple proportionality effects are expected.

The high turbidity levels released by dredging operations clearly have the potential to be deleterious to both phytoplankton and zooplankton communities. Very few field studies have observed significant direct effects on plankton production in the Great Lakes (IWGACPDA 1975; Sly 1977). Previously discussed difficulties and the relatively small area of impact in proportion to the open-lake plus the transience of the event could explain the absence of detectable effects. This assessment may not be valid in confined nearshore areas harbouring local or endemic populations, especially if either operation coincides with a sensitive or important life stage (Sly 1984). But this suggestion cannot be evaluated with the available data.

Phytoplankton are known to be effective bioaccumulators of contaminants.

The majority of species including diatoms, green, and blue-green algae, accumulate pollutants by adsorption (Conway and Williams 1979; Denny and Welsh 1979; Hassett et al. 1980). The bioaccumulation of many compounds varies with the plankton species and the compound in question. Hassett et al. (1980) showed that *Chlamydomonas* sp. was more efficient at accumulating Cd, Pb, and Hg than either *Nostoc* sp. or *Oscillatoria* sp. Keeney et al. (1976) demonstrated that *Cladophora*, a common benthic algal species in the Great Lakes, accumulated Zn, Cd, Pb, and Cu. *Fragilaria crotonensis* and *Asterionella formosa* accumulate Cd as a function of its concentration in solution (Conway and Williams 1979). Conway (1978) found that *A. formosa* also adsorbs As in amounts dependent on its ambient levels, but not HCB (Lederman and Rhee 1982a). Lead body burdens in a marine alga, *Platymonas subcordiformis*, varied with the concentration in the water (Hessler 1974).

Many chlorinated organic compounds also accumulate in algae via simple absorption or partitioning (Sodergren 1968; Reinert 1972). Lederman and Rhee (1982a) studied *F. crotonensis*, *Ankistrodesmus falcatus*, and *Microcystis* sp. and found that surface adsorption contributed only slightly to the accumulation of HCB, which varied with the species. They also demonstrated that HCB uptake was similar in live and dead cells. Dieldrin accumulation in the alga *Scenedesmus obliquus* was directly proportional to the concentration of dieldrin in the water (Reinert 1972).

The majority of studies have reported that contaminants have an inhibitory effect on phytoplankton productivity. Wong et al. (1978) examined the effects of ten metals (As, Cd, Cr, Cu, Fe, Pb, Hg, Ni, Se, and Zn), individually and combined, on phytoplankton cultures and natural phytoplankton assemblages in the Great Lakes. They found that the individual metals were not toxic at water quality objective levels recommended by the IJC, but when these metals were combined, the primary productivity of both the cultures and natural assemblages were inhibited. Nannoplankton (<64 um), particularly micro-algae and ultraplankton (<10 um), were consistently more sensitive to inhibition by heavy metals than net plankton (>64 um; Munawar 1982). Munawar et al. (1983), using C^{14} algal fractionation bioassays, found that heavy metals released from Niagara River sediments reduced photosynthesis 40-50% in micro-algae (<5 um), 10-90% in ultraplankton (5 to 20 um), and 20-80% in microplankton/net plankton (>20 um) as the dosage of a standard elutriate increased from 5 to 40%. Similar work by Munawar et al. (1985) demonstrated the differential sensitivity of algal size fractions to sediment-associated heavy metals in the Detroit River. Ultraplankton (<20 um) carbon assimilation was reduced 25% as the elutriate dosage increased to 20%. Microplankton/net plankton (>20 um) productivity was also inhibited, but not to the same extent as the smaller size fraction.

Phytoplankton productivity is not necessarily inhibited by contaminants. Lederman and Rhee (1982b) found that the rates of cell division and photosynthesis per unit of chlorophyll a in P-limited Fragilaria crotonensis cultures initially increased with the bioaccumulation of hexachlorobiphenyl (HCB), but declined sharply when cellular HCB concentrations exceeded approximately 2.0×10^{-7} ng/cell. In contrast, Ankistrodesmus falcatus and Microcystis sp. cultures exhibited no such effects at comparable cellular and external concentrations of HCB. Gotham and Rhee (1982) reported that photosynthesis was stimulated in A. falcatus and Melosira sp. cultures at average cellular HCB concentrations of 3×10^{-5} and 6×10^{-6} ng/cell respectively. However, cellular concentrations of pentachlorophenol (PCP) two to three orders of magnitude lower than HCB appeared to inhibit photosynthesis in A. falcatus and Microcystis sp. Few studies have focused on the stimulatory effects of contaminants even though such effects can be as ecologically destabilizing as the inhibitory effects (Rhee 1982). The underlying mechanisms responsible for the disruption of normal cell division and the stimulation of photosynthesis are not clear (Rhee 1982), although the effects appear to be chemical and species-specific.

The responses to contaminants exhibited by the various phytoplankton size fractions are important ecologically. Nannoplankton (<64 um) account for more than 50% of the phytoplankton biomass and carbon fixation in the Great Lakes (Munawar and Munawar 1981). More importantly, the micro-algae and ultraplankton fractions, which constitute a major food item of herbivorous zooplankton (Ross and Munawar 1981), are extremely sensitive to contaminants such as heavy metals. Therefore, changes in phytoplankton productivity, particularly in the smaller size fractions, will affect the zooplankton assemblage, but to what extent is not clear.

Although zooplankton are an important component of most aquatic food webs, comparatively little information exists concerning the productivity effects of the majority of contaminants. Bioassays by Magnuson et al. (1976) suggest that Hg and Zn may affect the behaviour of Pontoporeia affinis and lead to increased predation. Further work is needed to confirm these results since the P. affinis samples were small and the experimental period was short. Borgmann et al. (1980) examined the effects of Cd, Cu, Hg, Pb, and As on the growth and mortality of copepods (Cyclops vernalis, C. bicuspidatus thomasi, and Diaptomus sp.) from the Burlington ship canal. Growth appeared to be most sensitive to Cd, Cu, Pb, and Hg accumulations in the late fall and winter and most resistant in early summer. Arsenic did not show a similar seasonal cycle. Poldoski (1979) found that the extent of Cd bioaccumulation in Daphnia magna from Lake Superior was affected by the presence of complexing agents such as humic acids, pyrophosphate, or aminopolycarboxylic acids. Biesinger et al. (1982) demonstrated that Hg bioaccumulation in D. magna was strongly dependent on its chemical form in the water. At about the same water concentrations (0.26 ug Hg/L) daphnids accumulated roughly 20 times more Hg when added as methyl $HgCl_2$ then when added as $HgCl_2$. This is noteworthy because inorganic Hg and Hg compounds can be converted to methyl Hg by microorganisms in the sediments (Jensen and Jernelov 1969). Huckabee et al. (1975) found that partitioning was of greater importance for the accumulation of methylmercury by Daphnia pulex than dietary uptake in a laboratory food chain study.

1.3.2 Aquatic macrophytes

The impact of dredging and disposal on freshwater macrophyte productivity has not received much attention. Despite the absence of suitable data, general comments in the literature seem to suggest that significant changes are unlikely, at least on a lake-wide basis. This is not true in shallow, confined areas that are relatively small compared to the entire lake. These areas (commercial harbours, river mouths, etc.) receive the majority of current dredging and disposal activity in the Great Lakes, and measurable effects on macrophyte productivity should be expected.

Mechanical removal at the dredging site and burial at the disposal site are the most obvious macrophyte effects. Productivity changes as a result will be directly comparable to the size of the area affected. Recolonization may take some time as the newly exposed sediments are generally somewhat unstable (Kaplan et al. 1974) and temporary declines in productivity can be expected. Recovery to predredging and disposal levels depends on the recolonizing species. Maintenance dredging may promote the recolonization of pioneer species adapted to frequent disturbances with concomittant changes in species diversity and productivity. Productivity changes due to confined diposal site placement may well be permanent.

Short-term increases in turbidity and nutrient levels probably have negligible effects. However, flocculent material resuspended at the dredging and disposal sites may settle on macrophytes in adjacent low energy areas and interfer with photosynthesis (Kaplan et al. 1974). The productivity of species such as Myriophyllum sp. which reproduce vegetatively could be enhanced by the fragmentation of stems and roots.

Aquatic macrophytes provide shelter and attract food organisms for a variety of fish (Jaworski and Raphael 1979; Simser 1982). To a certain extent, moderate increases and decreases in macrophyte abundance may be beneficial to different components of the fish community. Extreme changes in abundance are not beneficial, however, the limited spatial extent of most dredging and disposal sites will likely have a mitigating influence on macrophyte disruption. Capital works projects and disposal in wetland areas can be expected to have the most serious impact on macrophyte communities.

1.3.3 Benthic invertebrates

The impact of dredging and disposal may be more severe on the benthic community than any other aspect of the aquatic ecosystem because of the relative immobility of most benthic organisms. The direct mechanical effects of the equipment used will inevitably disrupt community structure and productivity at the dredge site (IWGACPDA 1975). The initial impact at the Bronte Harbour dredge site was large scale destruction and/or export of benthic animals (Chemex 1973). The total number of animals declined from $29000/m^2$ to $4100/m^2$ two weeks after dredging. This represents a decline of about 85 percent.

Burial of benthic organisms and habitat at the disposal site is the most obvious effect (Kaplan et al. 1974; Bertrand et al. 1976; McCauley et al. 1977; Flint 1979; Allen and Hardy 1980). Maurer et al. (1980) found that the mortality of three estuarine molluscs (*Mercenaria mercenaria*, *Nucula proxima*, and *Ilyanassa obsoleta*) increased with increased sediment depth, burial time, and overlying sediments whose particle size distribution differs from the native sediments. Associated with open-water disposal is the creation of a thin layer of "fluid mud" which rapidly moves outwards from the centre of impact at the disposal site. This high suspended solids, low dissolved oxygen layer is not dense enough to support the weight or activity of the benthic animals it blankets (Peddicord 1980). Since upward mobility is restricted many organisms perish.

The available evidence suggests that the impact on benthic communities subjected to frequent maintenance dredging is only temporary and that repopulation is rapid. McCauley et al. (1977) found that the benthic infauna of Coos Bay, Oregon, readjusted to predredging conditions within a month in dredged areas and within two weeks in adjacent areas. They suggested that the benthos were well adapted to survive maintenance dredging because they were normally subjected to frequent disturbances from ship movements and other harbour activities.

The effects of silty mud from Cleveland Harbour on the benthos at an open-water disposal site in Lake Erie were examined by Sweeney et al. (1975). Changes in the biota after dumping were noticeable for at least five years after disposal had occurred. Flint (1979) compared the effects of open-lake disposal of coarse sandy sediments from the Ashtabula River and silty muds from Ashtabula Harbour. Although the effects were of shorter duration than observed by Sweeney et al. (1975), populations of isopods, nematodes, pelecypods, and gastropods declined at the dumpsites. He noted that opportunistic species, particularly oligochaetes, were able to take advantage of disturbed sediments and rapidly recolonize the disposal sites. The dominance of these species and larger population fluctuations than at the control sites were considered indicative of an unstable community structure. Flint concluded that more than a year was needed for the benthic community in disposal areas to reestablish a structure similar to unaffected areas. Reestablishment of the benthos at a Green Bay disposal site depended on the stability of the spoils and the level of contamination (Bertrand et al. 1976). "Cleaner" sediments were recolonized much faster than highly polluted sediments typical of many harbours. Spoils placed in shallow, high energy wind/wave environments were less likely to be recolonized because they were constantly reworked and unstable.

High turbidity levels could indirectly affect benthic organisms. The resuspended sediments can clog gills and impede the efficiency of filter feeding animals (Wright 1978). Laboratory studies by Peddicord and McFarland (1978), however, found that suspended sediment levels far in excess of those normally expected at dredging and disposal sites were tolerated without discernible effects by a variety of freshwater benthic species.

The greatest concern is the potential for contaminant bioaccumulation and biomagnification. Benthos are likely to have high body burdens because they are in constant contact with sediments and those that filter feed may be exposed to large volumes of contaminated water and particles. Three Illinois River clams (*Fusconaia flava*, *Amblema plicata*, and *Quadrula quadrula*) accumulated varying levels of Cu, Ni, Pb, Cr, Li, Zn, Co, and Cd (Mathis and Cummings 1973). Tissue concentrations generally exceeded concentrations in the water, but were less than those in the sediments. The data given probably overestimate most metals because the whole clam including the shell, intestines, and undigested gut contents, were used in their analyses. In contrast, tubificids from the same river accumulated metals at concentrations similar to those in the sediments. Namminga and Wilhm (1977) found that concentrations of Cu, Pb, and Zn in chironomid larvae exceeded sediment levels while Cr was somewhat less concentrated in the midges compared to the sediments. They indicated that Zn accumulations occurred via surface adsorption and that frequent molting resulted in the elimination of adsorbed metals.

High tissue concentrations of contaminants in benthos cannot always be correlated with elevated levels in the sediments or water (Neff et al. 1978). Neff et al. examined the bioavailability of sediment-adsorbed heavy metals from three harbours to Rangia sp., Palaemonetes sp., and Tubifex sp. Only 36 out of a possible 136 metal-organism-sediment combinations demonstrated significant metal accumulation and in many cases the uptake was quantitatively marginal. Long-term exposure experiments showed that significant accumulations of individual metals occurred in at least one test species. Short-term tests were less conclusive.

Very little is known about the bioaccumulation of organic compounds in freshwater benthic organisms. Most of the available studies have been reviewed by Munawar et al. (1984). Much more is known about bioaccumulation in marine invertebrates, but most of the work has focused on chlorinated organics. Nearly every animal tested in both field and laboratory studies has been shown to be capable of accumulating PCBs from the sediments.

It is difficult to assess the impacts of dredging and disposal on benthic productivity because few field studies have documented predredging conditions and biomass changes. The available information suggests that productivity changes can be expected as a result of removal from the dredge site and burial at the disposal site rather that chemical toxicity (Wright 1978). The long-term effects of high body burdens of contaminants are not known with certainty.

The magnitude of productivity changes will depend on the bottom area covered. At deep-water dumpsites declines may last one to five years depending on the frequency of use. In shallower areas subjected to the reworking action of wind and waves, recolonization may be delayed for considerably longer or prevented altogether. Confined disposal will probably result in a permanent loss of productivity. In contrast, changes at the dredge site seem to be transitory, lasting less than a year. All of these potential impacts could be severe if either operation coincides with a critical life stage or seasonal phenomena.

Although recolonization is expected to occur after dredging operations cease, the resulting community structure and function may be completely different from the predredging community. This could have both positive and negative effects on productivity depending on the species involved. Few studies with sufficient temporal scope are available, consequently these statements cannot be tested.

1.3.4 Fish

Concerns about the impacts of dredging and disposal on fish populations include turbidity, nutrient enrichment, dissolved oxygen depletion, increased contamination and toxicity, and changes related to the productivity of other biotic components of the system. Many of these concerns have been covered in previous sections. Fish populations that are concentrated in small areas for spawning, feeding, or rearing purposes are highly vulnerable to disturbances (Morton 1977). The need to protect these critical areas is great since the vast majority of current dredging and disposal occurs in the nearshore areas frequented by fish.

Increased turbidity levels are the most visible effect of dredging and disposal. Excessive turbidity levels characteristic of intensive dredging activity have not be shown to cause acute physiological problems or mortality of fish (IWGACPDA 1975); nor is there evidence of gill clogging and abrasion or impaired feeding and excretory functions. Laboratory studies suggest that the eggs, larvae, and adults of marine and freshwater species are capable of tolerating suspended sediment concentrations as high as 500-1000 mg/L (Sweeney 1978; Messieh et al. 1981) although there is some evidence of reduced hatching success in marine herring (Messieh et al. 1981) and slower growth rates in freshwater species (Bierman and Swain 1978).

While high turbidity is not directly lethal, migrating fish may be disoriented by the confusing mixture of different organic smells or by physical disturbances such as noise (Allen and Hardy 1980). This may interupt or prevent spawning and feeding migrations. Veshchev (1981) noted a significant mortality of migrating fry and larvae of two sturgeons, *Acipenser guldenstadti* and *A. stellatus* associated with dredging in the Volga River. The natural seasonal migrations of inshore species such as smallmouth bass, crappie (*Pomoxis* sp.), and yellow perch were restricted by turbidity plumes from dock construction at Nanticoke, Lake Erie (Chamberlain 1976). These studies strongly suggest that the timing and location of dredging and disposal activities are critically important. If they occur in the same location annually, then significant productivity effects can be expected because of altered migration patterns. This is particularly true of operations near the mouth of important spawning streams or similarly important feeding and nursery locations.

Dredging and disposal operations may physically disturb spawning shoals, feeding sites, and structures providing cover as well as destroying fish eggs, benthos, and other sessile or slow moving animals located at both the dredging and disposal sites (IWGACPDA 1975; Acres 1983). In addition, resuspended sediments may settle on spawning shoals and macrophyte beds near the dredging and/or disposal site. Most of the above effects are probably of greatest significance to the younger age classes of fish; adult fish are less likely to suffer directly because they can avoid the affected areas. Nonetheless, if dredging or disposal operations occur during critical life history periods (i.e. spawning, incubation, hatching), then significantly reduced year class strengths and recruitment to adult age classes can be expected. However, whether the impacts on fish production are temporary or permanent depends to a large degree on the frequency and extent of the area affected plus the availability of other suitable, but unaffected areas nearby.

The disruption of spawning areas (physically and by settling sediments) is potentially the most serious problem, particularly as it affects fish stocks that rely on specific shoals or nearshore locations for most of their reproduction. Lake whitefish, lake trout, and possibly lake herring are the species of particular concern in this respect. The former two are known to have formed a number of discrete stocks in the Great Lakes which migrate to specific shoals and (or) shoreline locations for spawning (Ihssen et al. 1981; Brown et al. 1981; Goodier 1981). This may be true of ciscoes (all Coregonus species except clupeaformis) also, but the evidence of stock discreteness is less convincing without more data (Todd 1981). The majority of salmonines and coregonines are unlikely to compensate for locally unsuitable spawning conditions quickly enough to ensure survival because the use of certain spawning locations is a genetic trait (MacLean and Evans 1981) or the result of imprinting.

The effects of dredging related nutrient enrichment and oxygen depletion on fish productivity are likely of little significance. Both conditions by themselves are temporary, extremely localized, and easily avoided by fish. However, increased oxygen demand in the hypolimnion may hasten anoxia, reducing the habitat available to coldwater species and subsequent production in confined areas such as harbours and enclosed bays.

Changes in planktonic and benthic productivity could affect fish productivity since both comprise substantial portions of the diet of many fish species. The extent of the impact will depend on the particular planktonic and benthic species affected and the availability of equally suitable organisms nearby. In addition, the ability of fish to adapt to new prey species will also play a prominent role. Changes in the food web are expected to affect larval and juvenile fish most severely because they generally rely more heavily on lower trophic level food resources than adult fish. Short-term changes (less than one year) are not expected to have a noticeable effect on overall fish production.

The remobilization of contaminants and toxic substances are probably the most serious problems associated with dredging and spoils disposal (Oster 1980; Acres 1983). Numerous field and laboratory studies, for example Reinert (1972), Eaton et al. (1978), Seelye et al. (1982), and Sodergren (1984) have demonstrated that fish can accumulate a variety of metals and organic compounds from many different sources. Studies reviewed by Hebert and Schwartz (1983) and Munawar et al. (1984) show that the route and significance of bioaccumulation and biomagnification depend on the compound and the species in question. Uptake through the food web is probably the single largest contributing factor in most situations since fish are usually the intermediate or top predators (Sly 1984). But tissue concentrations cannot always be correlated with contaminated food sources. Olson et al. (1973) found that the uptake of methylmercury directly through the gills and skin were important in rainbow trout and Seelye et al. (1982) demonstrated that yellow perch could accumulate several common contaminants including Zn, Fe, Hg, Cs, Se, and PCBs directly from resuspended sediments.

Fish will rapidly recolonize disturbed and contaminated excavation and disposal sites (Sweeney 1978). This could be problematical on a short-term basis because release rates from the sediments may be relatively high. As a result, the risk of acquiring high body burdens of pollutants may be much greater than normal. The long-term effects of elevated tissue concentrations are not known with certainty. Fish health studies have linked contaminants as a contributing factor to a variety of cancers and diseases. But the effects on reproduction, growth, and stock or community production are not clear.

There is evidence that the long-term impacts of confined disposal may be a greater problem than originally anticipated because of difficulties with site design and maintenance that permit the overflow of small particles containing sorbed contaminants (Oster 1980). Leaching, dehydration, and methylation of metals from these sites have been implicated as mechanisms through which long-term contamination of fish may occur (Sly 1977).

Although the impacts of dredging and disposal on fish productivity have not been explicitly evaluated, it is reasonably certain that they are site-specific. Since there is generally a loss of habitat diversity, some alterations in fish community structure and function should be expected near both the excavation and disposal sites. Observations at Port Stanley indicate that pelagic fish altered their distribution locally, near the dredge site (IWGACPDA 1975). Similarly, changes in the benthic and planktonic communities will have varying effects depending on the availability of other food items. In contrast, the available evidence suggests that water quality changes will probably have minimal effects depending on the timing and location.

1.4 Summary of dredging and disposal impacts

Table 3 summarizes the probable effects of dredging and spoils disposal on the productivity of aquatic ecosystems. It shows the potential impact that changes in physical, chemical, and biological variables may have on the productivity of the four main biotic components of most systems. Many of these impacts are short-term, causing temporary changes in productivity that may be difficult to detect. Others, such as contaminants, have long-term implications, but little specific information exists. Inferences from the short-term studies suggest that the long-term effects are probably negative in nature.

The chemical, physical, and biological effects of dredging and spoils disposal are site-specific and generally limited to the area near the excavation and dump sites. Most sites display a combination of effects and only rarely

Table 3. Summary of the effects of dredging and disposal on the productivity of aquatic freshwater ecosystems.

Adverse effects associated with dredging and disposal	Effects on Productivity			
	Plankton	Macrophytes	Benthos	Fish
1. Physical disruption of the bottom substrate	0	-	-	-
2. Turbidity[a]	$\underline{0}$	$\underline{0}$	0	$\underline{0}$
3. Contaminant release and remobilization	-[b]	0	-[c]	-[c]
4. Nutrient recycling	+	$\underline{0}$	$\underline{0}$	±
5. Settling sediments	$\underline{0}$	-	-	$\underline{0}$
6. Reduced dissolved oxygen levels	0	0	-	$\underline{0}$[d]
7. Disposal in littoral sites	$\underline{0}$	-	-	-
8. Disposal in deep areas	0	0	-	$\underline{0}$[e]

Note: This table does not show the effects of changes in the productivity in lower trophic levels on higher trophic levels.

+ Positive effects, i.e. productivity enhanced.
0 No discernible effect on productivity.
- Negative effects, i.e. productivity is reduced.

a - Reduced light penetration. Productivity effect depends on timing, location, and length of period involved.
b - Applies to phytoplankton and zooplankton.
c - Not known with certainty, but the evidence strongly suggests negative effects.
d - Oxygen depletion is not expected to have an impact unless it occurs after stratification in the hypolimnion or other regions where water column mixing is restricted. Oxygen may be a significant factor in interstitial water on spawning shoals.
e - Possible negative impacts on deep dwelling species such as sculpins as a result of buried habitats.

will a dredging operation elicit all the changes shown in Table 3. Although these changes are insignificant in the context of the Great Lakes, they are probably of greater significance in confined port and harbour areas, enclosed bays, and river mouths.

2. SEDIMENT CHARACTERISTICS AND INPUTS TO HAMILTON HARBOUR

Since many of the environmental problems in the harbour stem from the historic and continuing inputs of contaminants, which now reside in the sediments, some knowledge of sediment characteristics and distribution mechanisms is necessary to evaluate the environmental impacts of dredging and spoils disposal operations. The following section provides describes the sediment characteristics and dynamics in Hamilton Harbour. Every effort is made to use 1980s data. When this was not possible the latest 1970s data were employed. Although these data may not yield precise estimates, they are probably correct to within an order of magnitude.

2.1 The study area

Hamilton Harbour (43°14'N, 79°51'W) is located at the western end of Lake Ontario and is roughly triangular in shape, 8 x 5 km (Fig. 1), with a mean depth of 13 m, a volume of 2.8×10^8 m^3 and an area of 22 km^2 (OME 1977, 1978). The harbour is connected to Lake Ontario by the Burlington Ship Canal which is approximately 820 m long, 107 m wide, and 9.5 m deep (Dick and Marsalek 1973). An extensive shallow water wetland to the west called Cootes Paradise (and occasionally refered to as Dundas Marsh), communicates with the harbour via the abandoned Desjardins Canal.

Several creeks (Indian, Grindstone, Spencer, Sulphur, Ancaster, and Redhill) drain a small watershed of 500 km^2 into the harbour (OME 1977). Natural flows and inputs from sewage treatment plants (STP) amount to 3.5×10^5

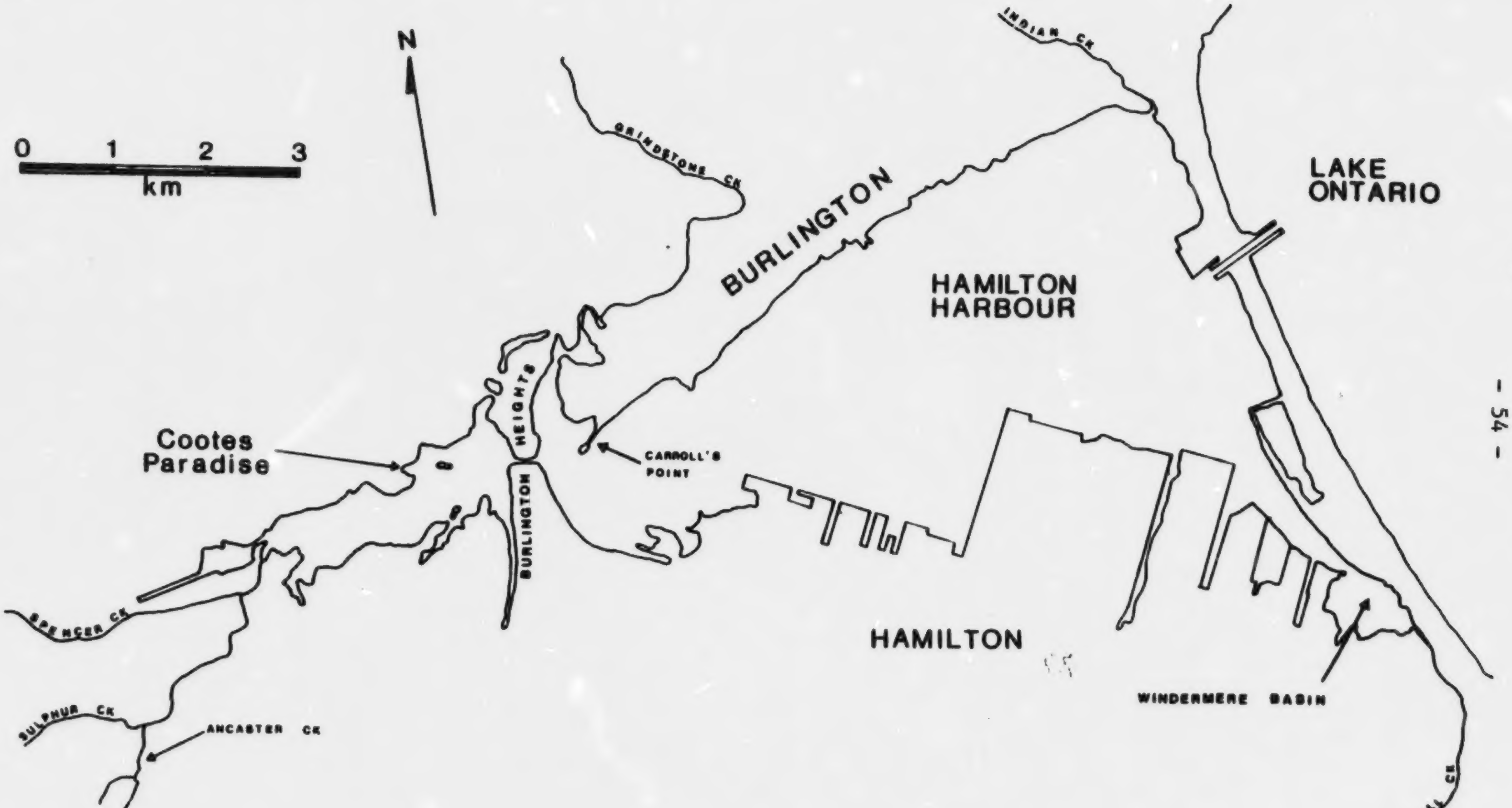


Figure 1. The Hamilton Harbour study area.

m^3d^{-1} and 2.8 x 10^5 m^3d^{-1} respectively (OME 1978) for a total throughput of 0.22% of the harbour volume per day (Kohli 1979). Untreated storm sewer overflows are estimated to contribute 8.6 x 10^3 m^3d^{-1} (Ng 1981) while 2.2 x 10^6 m^3d^{-1} is recycled by the industries along the south shore (OME 1978).

Exchange flow with Lake Ontario dominates the hydraulic balance of the harbour. Kohli (1979) calculated that an average of 2.04 x 10^6 m^3d^{-1} of harbour water flows into Lake Ontario and 0.73 x 10^6 m^3d^{-1} flows the other way, accounting for a net exchange rate of 0.5% of the harbour volume per day toward the lake. This is a rough estimate because Kohli based his calculations on certain simplifying assumptions of oscillatory flow and thirteen days of observations during September 1975. However, it generally agrees with earlier calculations by Palmer and Poulton (1976) who estimated a rate of 1.0% per day towards the lake using ten days of current meter recordings from September and October 1973.

The harbour has been identified by the International Joint Commission as an area of concern for priority cleanup since the early 1970s. The sediments are contaminated with Cu, Cr, Ni, Zn, Pb, Cd, Mn, Fe, and Co, nutrients, PCBs, and a variety of other organic compounds (OME 1977, 1978). The hypolimnion experiences severe oxygen depletion by late spring (Ng 1981) because of municipal and industrial discharges, polluted sediments, and algal decay (Great Lakes Water Quality Board 1983). Further water quality deterioration is partially prevented by the short hydraulic residence time of 0.25 years (OME 1985). However, the harbour is a chemical source with unknown impact to western Lake Ontario (Nriagu et al. 1983).

2.2 Sediment sources

Sediment inputs to the harbour are derived from a variety of point and non-point sources. Point sources include the Hamilton and Burlington STPs, Stelco and Dofasco. Non-point sources include Cootes Paradise, Redhill, Grindstone, and Indian Creeks plus several smaller streams along the north shore, and urban stormwater from Hamilton and Burlington. Although Cootes Paradise receives inputs from both point (Dundas STP) and non-point sources (storm sewers, Spencer Creek and other streams), it is considered a non-point sediment source in this report. The locations of the major contributors are shown in Fig. 2. Storm sewer locations were taken from Robinson and James (1982) while industrial and STP outfall locations are from OME (1981).

The older section of Hamilton, below the escarpment, is serviced almost entirely by a combined sewer system that drains directly into the harbour through 20 major outfalls during periods of high flow (Robinson and James 1982). The Regional Municipality of Hamilton-Wentworth is currently in the process of redirecting storm sewer overflows to the Hamilton STP to reduce untreated sewage loadings into the harbour (OME 1985). The storm sewer input to Cootes Paradise originates from sewers draining Dundas and four Hamilton sewers; three drain directly into the marsh and the fourth empties into Chedoke Creek.

The Hamilton sewers were ranked by Robinson and James (1982) in terms of estimated mean annual flow and loadings of suspended solids, BOD, total N, and P. They found that the top three sewers maintained the same ranking in every category. These sewers and their rankings are indicated by the numbers in Fig. 2. Combined, they account for an estimated 65% of the yearly flow and 63% of

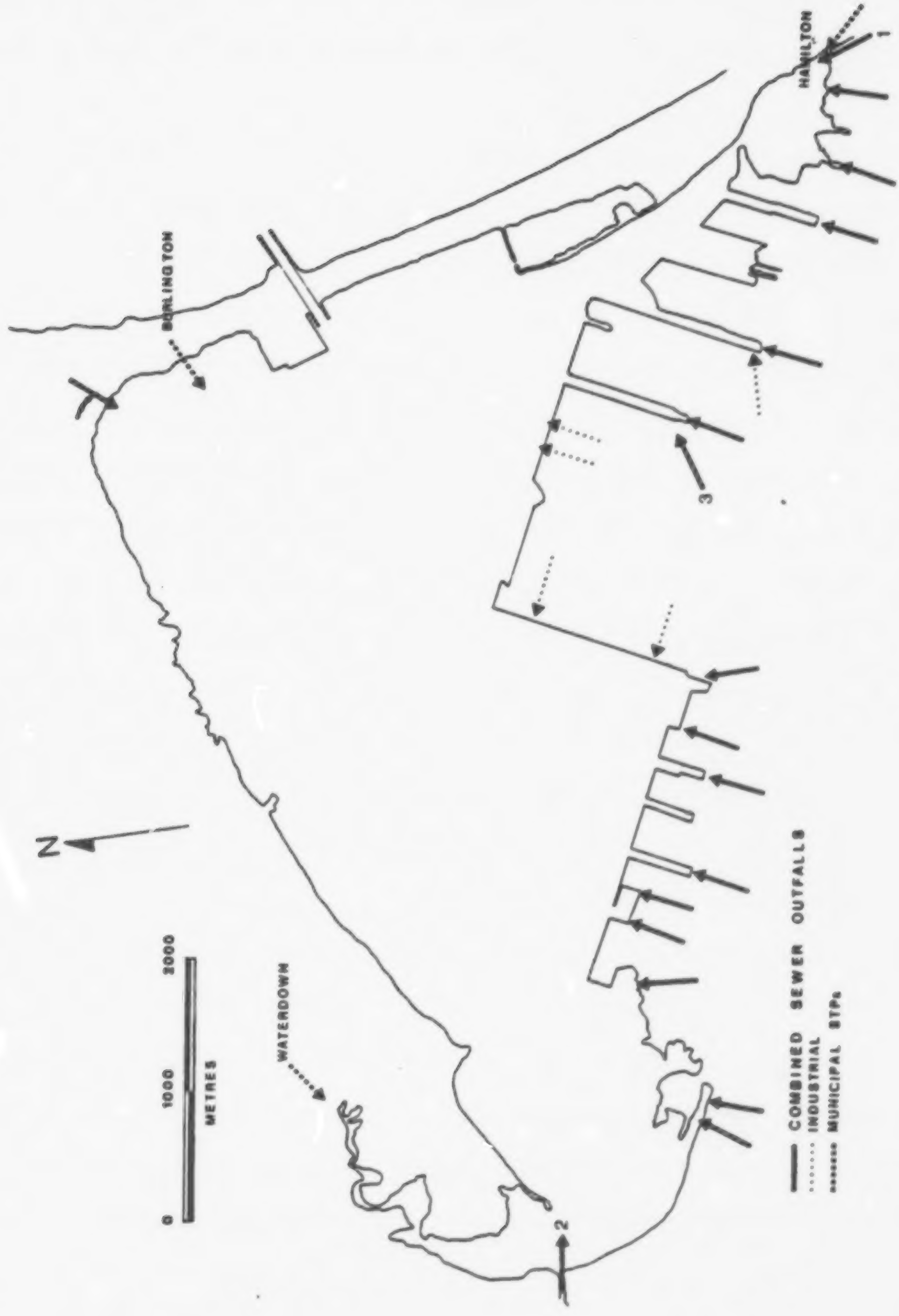


Figure 2. Point source inputs to Hamilton Harbour.

the annual nutrient and suspended solids loadings attributed to Hamilton stormwater runoff. The number two ranking belongs to the Chedoke Creek sewer in Cootes Paradise.

The harbour also receives treated effluent directly from two STPs (Hamilton and Burlington) and indirectly from two others (Dundas via Cootes Paradise and Waterdown via Grindstone Creek). The Hamilton STP accounts for 76% of the treated sewage effluent entering the harbour (Forde 1979). Most of the industries began discharging effluent directly to the Hamilton STP after its completion in 1963 and the rest were added in the late 1960s after the main interceptor sewers were built (OME 1985). Both Stelco and Dofasco presently discharge a portion of their wastes directly into the harbour, after treatment, through five sewers (four from Stelco and one from Dofasco).

Table 4 compares daily loadings of suspended solids for 1977 and 1982. Some 1983 data are also shown, but they are incomplete and lack resolution because monitoring efforts were not as extensive as in previous years. Between 1977 and 1982 suspended solids inputs to the harbour declined 66% and the percentage contribution from STP's and industrial sources increased from 9-12% and 55-65% respectively. Most of the decline in loadings is attributable to lower water flows particularly from industrial sources and Cootes Paradise. The increased flows from the STPs indicate that the efficiency of solids removal has increased. The Hamilton STP, Stelco, and Dofasco are currently the largest sources of suspended solids in the harbour.

The data in Table 4 are only provisional estimates of the loading rates. The figures vary from year to year depending on when and where the measurements

Table 4. Estimated suspended solids loadings to Hamilton Harbour.

Source	1977		1982		1983
	Flow	SS	Flow	SS	SS
Municipal					
Hamilton STP	253.0	11600	290.0	4930[b]	3730
Burlington STP	56.8	910	78.1	720[b]	
Industrial[a]					
Stelco	2.9	46000	-46.0	14500[b]	25300
Dofasco	-1.0	30200	-13.0	16300[b]	
Stormwater[c]					
Hamilton	21.8	1240	21.8	1240	
Burlington[d]	41.2	1500	41.2	1500	
Other Sources					
Redhill Creek	83.4	7600	59.4	1080[e]	
Grindstone Ck[c]	82.7	3300	82.7	3300	
Cootes Paradise	345.0	38000	95.6	3860[e]	
Total	**885.8**	**140350**	**609.8**	**47430**	**29030**

Note: Data taken from OME (1985). All flows x 10^3 m^3/d and loadings in kg/d.

a - Most industrial water is recirculated to the harbour. Net flows should be close to zero. Values different are measurement error. Small negative values may be due to evaporation by quenching processes and consumptive loss.

b - Calculated from data supplied by industry or STP.

c - Only 1977 data available.

d - Figures include several minor streams along the north shore.

e - 1979 data.

SS - Suspended solids.

were made. For example, inputs from Cootes Paradise (primarily due to Spencer and Chedoke Creeks) are particularly evident during spring runoff and after heavy rainfalls (M. Zarull, Ont. Min. Environ., Toronto, pers. comm.). They also assume that steady-state conditions prevail in that loading rates do not change on a seasonal basis. Grindstone Creek and storm sewer data were only available for 1977 and similarly, the latest sediment loadings for Redhill Creek and Cootes Paradise are from 1979. In addition, Robinson and James (1982) suggest that storm sewer figures should be doubled to account for all possible sources of error. It is assumed in all cases that no further changes have occurred in the 1980s.

A tentative suspended solids budget was drawn up for the harbour using existing data and a mass balance approach (Table 5). Atmospheric inputs were assumed to be negligible. Strictly speaking this may not be a valid assumption, but none of the easily accesible data have documented the atmospheric inputs of solids from the heavy industries in Hamilton. The relatively good agreement between total inputs and outputs (within about 10%) is noteworthy considering the errors inherent in the loading and sedimentation estimates. The annual amount sedimenting out over the entire harbour is probably overestimated because it is based on sediment trap data and therefore includes organic matter settling out of the water column (M. Charlton, Nat. Wat. Res. Inst., Burlington, pers. comm.). Table 5 shows that roughly 85% of the total suspended solids load entering the harbour from all sources are retained in the basin. The remaining 15% is exported to western Lake Ontario.

Table 6 compares daily loadings of trace metals from all sources in 1977 and 1982. Some trends are evident, especially a reduction in Fe from both steel

Table 5. Estimated annual suspended solids budget for Hamilton Harbour.

INPUTS[a]		OUTPUT		% of Harbour Input
STP	4.6	To Lake Ontario	5.8[b]	15
Industrial	27.8	To sediments	52.0[c]	85
Stormwater	1.0			
Others[d]	17.8	Total	57.8	
Total	51.2			

Note: All data are in 10^6 kg/y and are from OME (1985).

a - 1977 data.

b - Export to Lake Ontario calculated for period May 1979 to April 1980.

c - Calculation based on unpublished data for sedimentation rate over the entire basin (M. Charlton, unpub. data cited in OME 1985).

d - Others include Redhill Creek, Grindstone Creek, and Cootes Paradise.

Table 6. Mean trace metal loadings to Hamilton Harbour in 1982.

Source	Fe	Cu	Zn	Mn	Cd	Cr	Pb
1977							
Hamilton STP	299	18	86	33	1.3	38	10.1
Burlington STP	7	1	5	5	0	1	0
Stelco	21500	110	990	350	0	31	15
Dofasco	13000	23	110	220	0	58	6.6
Redhill Creek	18	0.8	2	8	0.42	1.7	2.5
Grindstone Creek	12	0.8	2	8	0.41	1.7	2.5
Cootes Paradise	1100	7.6	23	88	1.8	7.0	10
Hamilton Runoff	80	0.7	9	8	0.1	1.1	1.5
Burlington Runoff	9	0.6	1	2	0.2	0.8	1.2
1982							
Hamilton STP[a]	230	9.5	21	24	<0.5	<9	<7
Burlington STP[a]	64	2.7	7.0	3.5	<0.2	1	<2
Stelco[b]	6300	8	214			27	27
Dofasco[b]	4030	0	73			0	4
Others[c]	290						

Note: All data are in kg/d and are from unpublished reports compiled in OME (1985).

a - Loadings are for the period May-Sept. 1982. Means are based on 12-14 samples.

b - Loadings are net loadings and are calculated from arithmetic mean concentrations without discarding outliers. Individual waste streams were sampled 2-4 times per year.

c - Other sources include Redhill and Grindstone Creeks, Cootes Paradise, and stormwater runoffs from Hamilton and Burlington. These data are for 1982.

mills and Zn from Stelco. The decrease in Zn is due to recirculation of blast furnace scrubber water (OME 1985). Loadings of Zn, Cr, Cu, Cd, and Pb from the Hamilton STP also declined by 1982. This is believed to be the result of reduced industrial loadings to the plant (OME 1985). Figures for Pb, Cd, and Cr from both STP's and Ni from Burlington in 1982 represent the detection limits since the actual values were somewhat less (OME 1985). Iron loadings from the Burlington STP increased slightly by 1982 due to the use of ferric chloride for phosphorus removal since 1976 (OME 1985).

Table 7 shows the annual loadings of trace organic compounds from municipal point sources in 1982-83. Loadings from the Hamilton STP are generally double those from Burlington. No data are available on loadings from industrial and non-point sources. However, Wang (1981, cited in OME 1985) found PCBs, organochlorine pesticides, PAHs, and chlorinated benzene in urban runoff from Hamilton during 1980. It was not possible to calculate loading rates because the sample sizes were very small (1-5 per group).

Approximately 40 PAHs were identified in the Hamilton STP effluent and specific waste streams from Stelco and Dofasco in 1978 (OME 1985). Concentrations in the STP effluent ranged from 0.02 to 0.3 ug/L. Higher concentrations were noted in coke plant effluents (1-24 ug/L), but because of dilution with cooling water final concentrations discharged to the harbour were similar to the STP effluent. Further sampling in 1982 showed that concentrations in the steel mill effluents were mostly less than 5 ug/L and many were below the detection limit of 0.1 ug/L (OME 1985). These data and others from MOE indicate substantial variability in concentrations that could not be

Table 7. Trace organic loadings to Hamilton Harbour during 1982-83.

Parameter	Hamilton STP	Burlington STP
Total PCB	4.6	0.9
α-BHC	0.5	0.1
β-BHC	0.4	0.1
γ-BHC	1.8	0.4
α-chlordane	<0.2	<0.1
γ-chlordane	0.3	<0.1
Mirex	1.9	0.6
DDT & derivatives	<1.5	<0.1
HCB	1.4	1.1
Dieldrin		<0.1

Note: Data from OME (1985). All loadings in kg/y. Loading data are calculated by multiplying the mean concentration of 36 samples by total annual flows: Hamilton STP - 1.06×10^8 m^3/y
Burlington STP - 2.85×10^7 m^3/y

resolved because of the small number of samples. Loadings were not calculated because reliable measurements were not available.

Many trace metals and organic contaminants are transported into the system by suspended particulate matter. Since more than 80% of the suspended solids are retained in the harbour, some knowledge of their chemical constituents is necessary when determining the fate of pollutants. Limited data are available from McIsaac et al. (1982) who examined the trace-metal constituents of suspended sediments from the southeastern portion of the harbour in 1981. These data are compared with suspended sediments from the deep basin (station 258) in Table 8. The differences between the two areas are not great and are explainable on the basis of spatial and temporal variability in the harbour (OME 1985). Table 8 shows that the suspended sediments have large contaminant concentrations.

McIsaac et al. (1982) fractioned the metals into five groups and assessed their ease of release from the sediments. Readily exchangeable metal ions comprised a significant portion of the total metal content for Cd (13%), Co (15%), Cu (15%), Mn (15%), and Ni (20%). The ions in this fraction are believed to arise from inputs of salt-laden water ($CaCl_2$) during spring runoff and are probably available to the biota during sensitive stages in the life cycle of many organisms. Carbonate and surface oxide bound forms were significant for Cd (24%), Co (11%), Mn (20%), Ni (8%), Pb (6%), and Zn (16%). Iron-manganese oxides were the most important phases regulating the chemical forms of the trace metals studied.

The total annual loadings of some metals and organic compounds to the harbour and sediments are shown in Table 9. These data were not available for

Table 8. Mean trace metal concentrations of suspended particulate matter in Hamilton Harbour.

Element	Southeast[a] 1981	Stn 258[b] 1982
Al (mg/g)	1.80 ± 0.49	
Cd	16 ± 3	
Co	18 ± 3	22 ± 9
Cr	265 ± 55	144 ± 51
Cu	140 ± 4	103 ± 30
Fe (mg/g)	2.95 ± 0.43	35 ± 9
Mn (mg/g)	8.07 ± 4.35	20 ± 9.5
Ni	110 ± 60	50 ± 14
Pb	365 ± 70	192 ± 86
Zn	1970 ± 530	1050 ± 227

Note: All figures are in ug/g unless otherwise indicated and show the mean ± standard deviation.

a - From McIsaac et al. (1982). Based on 34 samples taken during July and August 1981 in the southeast portion of the harbour and the Windermere channel.

b - From Poulton (1983 shown in OME 1985). Based on 6 samples taken from station 258 in the deep central basin of the harbour. See Fig. 3 for station location.

Table 9. Total contaminant loadings from all sources to Hamilton Harbour and its sediments.

Parameter	To Harbour[a]	To Sediments OME[c]	 Other[b]
Cu	58	2	1.8
Zn	438	29	45
Pb	18	3	5.9
Fe	13100	760	880
Mn	263	110	18
Cr	51		4.2
PCB	50 kg/y[d]	23 kg/y	
α-BHC	5.1 kg/y[d]	<0.1 kg/y	
α-chlordane		0.3 kg/y	
γ-chlordane		0.4 kg/y	

Note: All figures are x 10^3 kg/y unless otherwise indicated. Metals data are from 1975 and organics data are from 1982-83.

a - Modified from OME (1977) by Nriagu et al. (1983).

b - From Nriagu et al. (1983).

c - From OME (1985).

d - Figures are approximations from OME (1985).

other compounds. Loadings to the sediments were estimated by multiplying the mean surface concentration by the total mass of sedimenting material (OME 1985). The average measured sedimentation rate in the harbour is 6.5 $g/m^2/y$, which is 5.2 x 10^7 kg/y over the entire basin (M. Charlton, unpub. data cited in OME 1985). These figures are from sediment trap data and do not account for settling of resuspended sediments.

Table 9 indicates that a significant portion of the metals and organic compounds entering the harbour are retained, but an even larger portions are exported to Lake Ontario. For example, the harbour is a sink for 23 kg/y of PCBs and exports approximately 24 kg/y. Some caution is needed when using the OME loading figures because the quality of the data was not adequately assured (OME 1985). Nevertheless, the sediment loading rates shown are unexpectedly low given that approximately 85% of the suspended solids are retained in the harbour. One possibility is that the majority of fine sediments entering the harbour (which generally have disproportionately large contaminant loads) are exported to Lake Ontario rather than settling in the harbour. This hypothesis could be tested by developing a detailed budget of the different sediment types.

The pollutant loadings data to Hamilton Harbour (Tables 4, 6, and 7) show that the Hamilton STP and the two steel mills are the major sources of suspended solids, iron, and several other heavy metals. Quantitatively, Cootes Paradise is also a significant source of suspended solids and metals. The loadings of most contaminants and suspended solids show a marked decline between 1977 and 1982.

Trace organics and PAH loadings from all sources could not be determined because only a small number of samples were analyzed for these compounds. The limited data indicate that urban stormwater runoff from Hamilton may be a source of PCBs and organochlorine pesticides. The two steel mills and the Hamilton STP are probably the major sources of PAHs.

Data from McIsaac et al. (1982) suggest that many of the metals many be carried into the system by suspended solids. Calculations by OME (1985) indicate that although the harbour is a sink for many contaminants, significant quantities are exported to Lake Ontario. These loadings are probably minor next to atmospheric and Niagara River inputs (OME 1985). However, the apparent lack of quality control on the loading data suggests the need for caution in all interpretations.

2.3 Sediment characteristics

The Ontario Ministry of the Environment has regularly monitored water and sediment quality in the harbour since 1975. Sediment samples have been obtained from 24 sites shown in Fig. 3. These sites will be referred to by their station numbers in the following sections.

2.3.1 Physical characteristics

The physical characteristics of the sediments have not received much attention. OME (1977, 1978) found that most areas had gray to black organic mud sediments. Silt and/or sandy-silt were present at all locations along the north shore and near the entrance to Cootes Paradise and the Burlington ship canal.

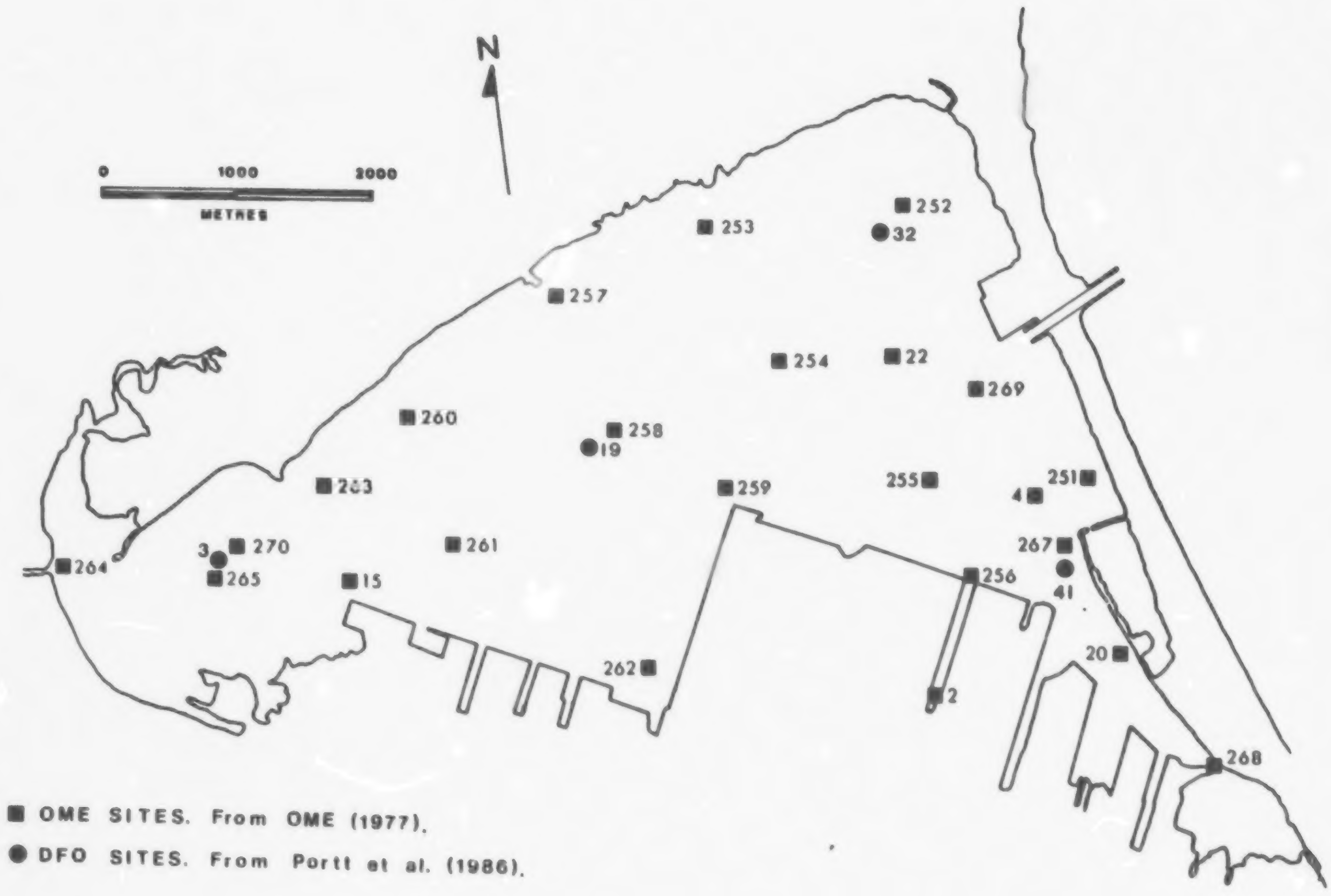


Figure 3. Sediment sampling locations in Hamilton Harbour.

Sediments south of the canal had a silty organic composition.

Only a limited amount of particle-size distribution data for the harbour sediments are readily available. Mudroch and Zeeman (1975) sampled an unspecified location and reported the sediment composition as 1% sand, 46.5% silt, and 52.5% clay. The results of more detailed analyses in 1977 and 1984 are shown in Table 10. These data emphasize the predominance of silt and clay in the harbour and show that the relative proportions of both size fractions increased between 1977 and 1984, especially at 258/19, which is located in the deep central basin. Stations 252/32 and 4/41 have slightly more sand and gravel than the other stations. This may be partly due to the influence of exchange flows through the Burlington Canal and partly to the slowing of strong currents as they enter the main body of the harbour; 252/32 is located off of the mouth of Indian Creek and 4/41 is located in the channel leading into Windermere Basin. The inconsistencies in the 1977 data for 252 were not explained by OME (1981).

Further data are needed in order to properly assess the impact of sediment disturbances by future dredging and spoils disposal activities in the harbour. These include depth distribution and contaminant analyses of the various size fractions. However, it is reasonable too suggest that the surficial sediments are easily disturbed because of their high silt and clay content. This is cause for concern because these size fractions tend to have proportionately higher contaminant concentrations than larger particles (Krantzberg and Bailey 1983).

Table 10. Comparison of particle-size distribution of the sediments in Hamilton Harbour in 1977 and 1984.

Station No.	Sample Date	Sediment Type			
		Gravel	Sand	Silt[a]	Clay[b]
270	1977	0.0	3.2	29.8	67.0
3	1984	0.0	0.3	40.9	58.8
258	1977	0.0	6.0	47.0	47.0
19	1984	0.0	0.5	27.9	71.6
252	1977	7.2	89.4	(-13)	17.0
32	1984	0.0	6.4	53.9	39.8
4	1977	0.1	11.4	32.1	57.0
41	1984	0.0	15.1	46.7	38.2

Note: All figures are percentages (%). The 1977 data were taken from OME (1981) and the 1984 data were taken from Portt et al. (1986). The 1986 stations shown are the ones closest to those sampled in 1977, but they may not coincide exactly.

a - Determined as the difference in 1977.

b - Determined by hydrometer in 1977.

2.3.2 Chemical characteristics

Changes in sediment quality between 1975 and 1980 are shown in Table 11. These data represent mean values for surficial sediments from all stations sampled in a given year combined. The sediments were considered heavily contaminated with Pb, Zn, Fe, Cr, P, and COD and moderately contaminated with Cd, Ni, Mn, Hg, and Cu in 1975 and 1976 (OME 1977, 1978). There are no clear trends, but Zn, Hg, and possibly Pb, Cd, and Co levels may have declined between 1975 and 1980. Regardless, all parameters still exceeded OME guidelines for open-water disposal in 1980 (see Section 3.2.2.1).

The highest nutrient and metal concentrations are located in the deep central basin and in areas adjacent to outfalls (OME 1978, 1985). Table 12 shows that sediments located next to major outfalls generally exhibit the highest concentrations of substances discharged in those areas. Windermere Basin sediments are contaminated with nutrients, PCBs, and heavy metals derived from the Hamilton STP and stormwater runoff from Redhill Creek.

Zones of distinctive sediment chemistry were defined by OME (1985) using factor analysis. The deep water depositional basin and areas adjacent to municipal and industrial outfalls along the south shore were similar in that they have the highest levels of contamination. These zones tend to have sediments consisting of predominately silt and clay. The central and southeast zones are affected by point-source discharges more than the less contaminated areas adjacent to the north shore and west end (OME 1978). Mass exchange of water through the ship canal (Kohli 1979) appears to be an effective mechanism maintaining higher sediment quality near the canal entrance and in the northeast

Table 11. Mean concentrations of contaminants in the surficial sediments of Hamilton Harbour, 1975–1980.

Parameter	Hamilton Harbour May 1975^{a}	1975^{b}	June 1976^{c}	May 1977^{d}	Oct 1977^{d}	Oct 1980^{d}	Windermere Basin^{e}
LOI^{f} (%)	10.5			9.1	9.5	9.8	2.5
BOD (mg/g)	3.9				2.6		
COD (mg/g)	152.0		140.0	150.0	150.0		
NH_3				30.0	10.0		
TKN (mg/g)	3.5	3.3	2.7	2.9	2.9	3.5	1.3
Total P (mg/g)	3.4	3.6	2.4	2.7	2.9	3.2	1.3
Fe (%)	6.8	13.3	7.0	8.8		6.8	4.2
Cu	101.0	135.0			110.0	130.0	500.0
Cr	206.0		200.0		220.0	204.0	1070.0
Ni	45.0		39.0		52.0	44.0	
Zn	3110.0	4160.0	2010.0	2600.0		2700.0	3400.0
Pb	320.0	1100.0	260.0	310.0	300.0	310.0	700.0
Cd	7.8		5.0		6.3	5.7	12.0
Mn	1670.0	2300.0				1900.0	
Hg	2.4		0.4	0.7	0.6	0.3	1.6
Co	15.0				12.0	10.0	
As				19.0	22.0		53.0
Al		3100.0					
No. of Samples	20	1	26	25	25	20	10

Note: All concentrations in ug/g (dry weight basis) unless otherwise noted.

a - From OME (1977).

b - From Mudroch and Zeeman (1975). Location was not specified.

c - From OME (1978). Mean results of top 5-7 cm of core samples.

d - From OME (1985). Results for October 1980 are means of top 3 cm of core samples.

e - From OME (1982). Means of 10 surface sediment samples from Windermere Basin.

f - LOI - loss on ignition or volatile solids.

Table 12. Highest sediment pollutant concentrations detected in shallow water areas of Hamilton Harbour between 1975 and 1980.

Contaminant	Station					
	268	Windermere Basin	20	2	256	262
Cu	340	800		400	180	
Cr	700	1800	500	1200	670	
Ni	80			190		
Hg	0.68	3.4	1.4	1.2	1.2	2.7
Pb	460	1100		1000		1200
Zn	2400	5200		8400		7800
Cd	7.6	18	8			12
As				60	50	
Fe[a]					170	140
Mn				4500	3200	2400

Note: Data from OME (1985). All figures are in ug/g (dry weight basis) unless otherwise indicated.

a - Concentrations in mg/g.

corner of the harbour (OME 1977, 1978).

The results of trace organic and PAH analyses of sediments collected in October 1982 are given in Table 13 (OME 1985). Median concentrations are shown for those organics detected in more than 50% of the samples. The sediments used for PAH analysis (sites 2, 20, 256, 258, 262, and 268 in Fig. 3) were selected for their proximity to suspected sources or depositional areas (OME 1985).

PCBs were the most commonly detected organic compound (85% of the samples). No spatial patterns were evident because of widely varying concentrations from one location to the next (OME 1986). The highest concentration (1270 ng/g) occurred at 257 which has historically had low pollutant concentrations (OME 1985). Most of the samples >500 ng/g were located in the deep central basin or along the south shoreline. Other commonly occurring organic pesticides were α-chlordane, γ-chlordane, β-BHC. Spatial distribution patterns for these chemicals were also not evident suggesting there is no obvious local source (OME 1986).

Maximum PAH concentrations were found at station 262 and minimum levels occurred at station 20 for all compounds (OME 1985). The order of stations from highest to lowest concentration for most PAHs is 262>2>258>268>256>20. This pattern reflects the input of PAHs from the two steel mills.

2.4 Sediment redistribution

The redistribution of sediments in Hamilton Harbour is governed by many factors including water currents, wind, shipping, and dredging. Some awareness

Table 13. Trace organics, PCBs, organochlorine pesticides, and PAHs in the sediments of Hamilton Harbour in 1982-83.

Compound	Mean (ng/g)	SE	Range (ng/g)	% of Samples Detected
PCB	440.	80.	ND-1270	85
Aldrin	<1.		ND-1	5
pp'-DDE	<1.		ND-9	25
op'-DDT	<5.		ND-10	15
α-chlordane	5.	1.	ND-18	55
γ-chlordane	7.	1.	ND-20	65
HCB	<1.		ND-4	25
α-BHC	2.	0.4	ND-6	50
β-BHC	2.	0.7	ND-13	15
PAHs[a]	Mean (ug/g)	SE	Range (ug/g)	
Fluoranthene	14.	6.	1.9-43.4	
Perylene	4.5	1.2	1.2-9.7	
Benzo(k)fluoranthene	3.8	1.1	1.1-9.0	
Benzo(a)pyrene	4.9	1.4	1.2-11.1	
Benzo(g,h,i)perylene	3.8	1.1	1.6-8.6	
Indene(1,2,3 cd)pyrene	3.9	1.2	1.1-9.7	

Note: Data from OME (1986). PCB's and organochlorine pesticides data are based on analysis of 20 samples taken in October 1982.

a - Based on one sample from stations 2, 20, 256, 258, 262, and 268. See Fig. 3 for station locations.

ND - None detected.

SE - Standard error.

of these interactions at the excavation and disposal sites is required when assessing the impact of dredging on sediments dynamics. Additionally, resuspension can enhance the remobilization of nutrients and contaminants into the system or transport them to relatively uncontaminated areas adjacent to dredging and spoils disposal sites. In the following sections each factor will be discussed in the context of Hamilton Harbour. Although quantitative estimates are desirable, they generally are not possible using the existing information base. It was beyond the scope of this study to obtain the new data required for such estimates.

2.4.1 Water currents

Circulation patterns and other water movements in lake basins develop as a result of wind shear (Sly 1978). These movements have the ability to selectively transport finer particulates of silt and clay already in suspension or resuspended by wave action (Sly 1978). The velocity of the currents and their patterns are of particular importance to sediment redistribution (Niimi 1982). High velocity currents are generally capable of carrying larger sediment loads which they dump when velocity decreases. Current velocities in the shore zone of the Great Lakes tend to be relatively high and fairly energetic compared to currents in the deeper waters which tend to be weaker (Csanady 1978). Consequently the deep areas of lakes are often permanent depositional basins for the sediments transported from nearshore areas by high energy currents. This point appears to be supported by the inferences drawn from the chemical characteristics of Hamilton Harbour's sediments (see Section 2.3.2).

Sedimentary processes in small lakes are dominated by river inflows and circulation (Sly 1978). Hamilton Harbour essentially acts as a small lake because of its nearly enclosed nature from Lake Ontario. Circulation in the harbour is complicated considerably by thermal stratification, large inputs of heated effluents, and fluctuating exchanges of epilimnetic and hypolimnetic water with Lake Ontario.

Although currents have been measured on several occasions between May and November 1972-1981 (OME 1985), very little velocity or pattern information is readily available. James and Eid (1976) measured surface current patterns on 12 different occasions under a variety of weather conditions and found that the currents were almost always downwind. Current patterns were influenced by wind speed, direction, fetch, water depth, and surrounding obstructions.

OME (1981) developed two-dimensional and three-dimensional computer models to simulate the currents in the harbour. The 2-D model was used to predict the currents when the prevailing wind was from the west at an average speed of 7.5 m/s. The model shows that the strongest currents would flow eastward with the wind along the north shore, following the shallow water around to the ship canal. South of the canal the results are confused and the current regime is difficult to interpret. A weaker flow with the wind was predicted for the western part of the south shoreline. The reduced current velocities were related to increased water depths. In the deep central basin a return current against the wind occurs with the strongest current being adjacent to the north shore current but in deeper water. Comparisons of observed and modelled currents showed significant variations (especially south of the ship canal) in velocity and direction probably because the 2-D model does not account for vertical exchange between the different water layers (OME 1981). Its usefulness, therefore, is restricted to non-stratified conditions.

To overcome the problem of turbulent vertical exchange the 3-D model was developed (OME 1981). The water column was divided into three layers and currents in each layer were modelled. This model shows that surface currents generally follow the wind direction with a slight deflection to the right due to the Coriolis effect. Mid-depth currents are deflected much farther to the right and bottom currents return against the wind. Predicted surface current velocities were the fastest while bottom velocities were the slowest. Model results when the wind is consistently from the west indicate a similar pattern to that predicted by the 2-D model. However, return currents against the wind are found in the mid- and bottom depths. Calibration of the model proved to be difficult, especially for currents in the mid-depths. The greatest problem appears to be limited vertical resolution which will require additional data as a function of depth to resolve (OME 1981).

A tentative plan of water circulation in Hamilton Harbour is suggested in Fig. 4. The current patterns shown were taken from Ozanian (1957) and the OME (1981) modelling studies. The prevailing wind direction is assumed to be from the west. Data from 1963-1980 indicate that winds from the southwest and west occur 40-65% of the time in a given month at the Hamilton Marine Police station (southwest region of the harbour) (Atmos. Environ. Ser. 1982). Current velocities were not estimated because no reliable data exists.

The pattern shown in Fig. 4 is probably complicated by flows through the ship canal. During unstratified periods flow is in the same direction at all

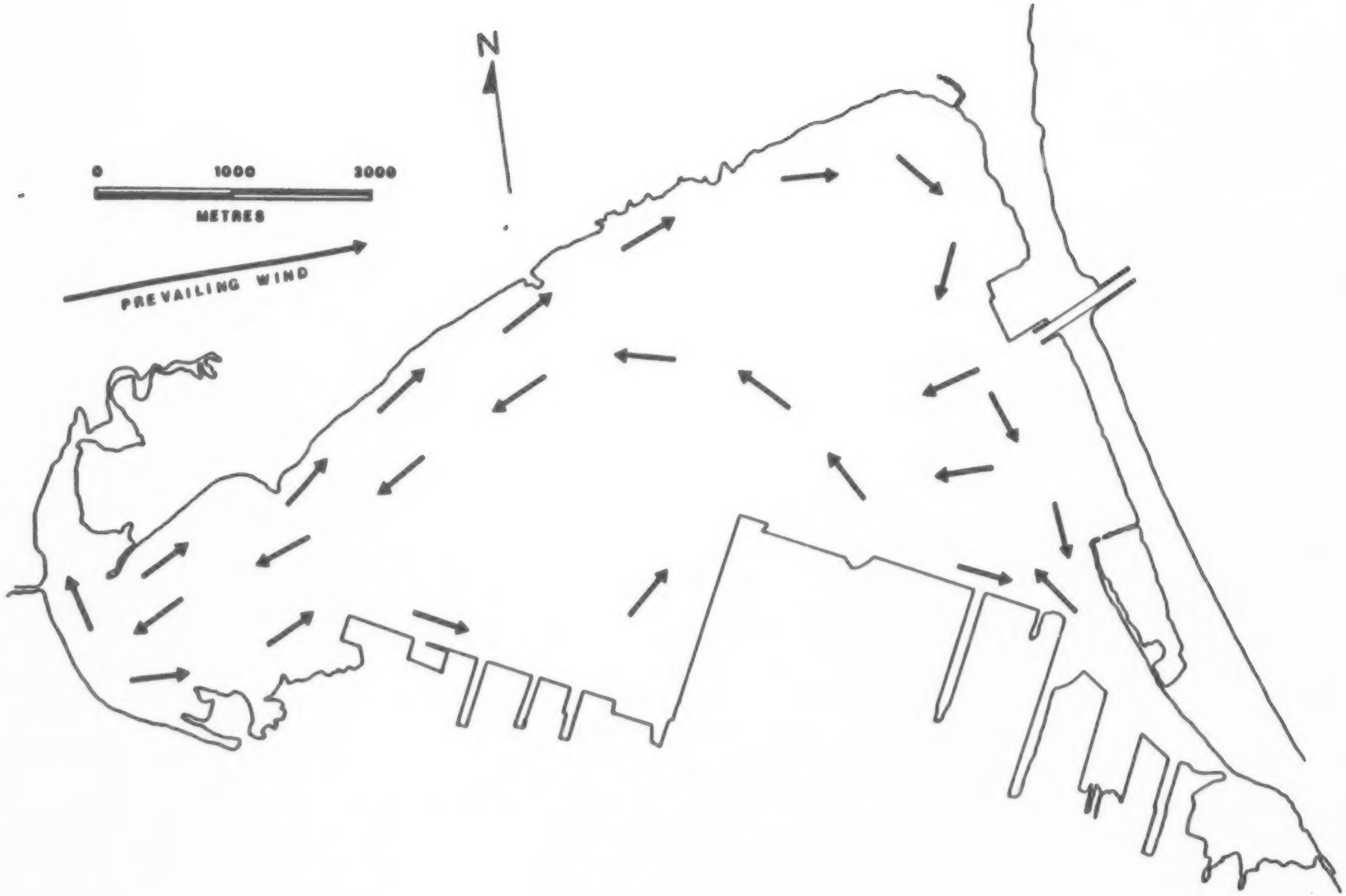


Figure 4. Water currents in Hamilton Harbour. Based on OME (1981).

depths while during stratification warm harbour water flows out on the top and cold lake water flows in on the bottom (Kohli 1979). In both cases flows oscillate back and forth. Measurements of these oscillations show that they have periodicities ranging from 20 minutes to half a day, although high frequency periods of 0.1-0.2 h have also been noted (OME 1985). These high frequency motions are probably responsible for some of the difficulties encountered by the modelling studies. They also induce a high degree of patchiness in thermal, chemical, and biological variables in the harbour (Harris et al. 1980a).

Assuming the circulation shown in Fig. 4 is a reasonable approximation of the actual currents, several points are evident. First, sediments transported by Grindstone Creek are probably deposited northwest of Carroll's Point. Second, sediment inputs from Cootes Paradise may be widely dispersed along either the north or south shore. In both instances deposition probably occurs towards the deep central basin. Third, the majority of sediments from Redhill Creek, the Hamilton STP, and various stormwater outfalls entering Windermere Basin probably end up in the approach channel to the Dofasco and Strathearne Avenue slips. Flows from Windermere Basin are generally directed towards the ship canal, and since the basin is nearly filled, most of the sediments are probably flushed out into the main body of the harbour. The rather frequent schedule of maintenance dredging in this channel in recent years (Section 3.1) supports this observation.

Much of the preceeding discussion is speculative in nature. A certain amount of reliance was placed in the predictions of the OME modelling studies even though serious difficulties were encountered in modelling currents near the ship canal and in the mid-depths. Resolution of these problems will require further model refinements (i.e. adding more water layers) and field data from drogue trackings in the harbour (OME 1981).

2.4.2 Wind

Wind is another potential sediment redistribution mechanism in the harbour. Wind resuspends the bottom sediments which are then transported by water currents. The effective fetch and water depth are two important factors governing sediment redistribution by wind (Sly 1978). The effective fetch is a measure of wind/wave generation in relation to the water surface over which the wind blows; as the fetch increases the wave base is lowered and the area of lake bottom subject to erosion and transportation increases (Hakanson 1981).

Hakanson (1981) noted that different regions of lake sediments are dominated by three processes: erosion, transportation, and accumulation. To determine the percentage of a given lake dominated by these processes he developed the following relationship based on three standard morphometric variables:

$$a_{E+T} = 52.0 \left| \frac{\sqrt{a}}{D_{max}} - 0.2 \right| x \frac{D_{max}}{3\bar{D}} x \log \left(\frac{60.6\bar{D}}{\sqrt{a}} \right) + 19.3$$

where a_{E+T} is the percent area dominated by erosion and transportation, a is the lake area (km^2), D_{max} is the maximum depth (m), and $\bar{D}$ is the mean depth (m). Applying this to Hamilton Harbour and subsituting $a=22$ km^2, $D_{max}=22$ m, and $\bar{D}=13$

m gives a_{E+T}= 20%. This implies that 20% of the total bottom surface area is dominated by erosion and transportation processes and that 80% is depositional area (a_A=100-a_{E+T}).

The critical depth separating areas of erosion and transportation from accumulation is derived using the hypsographic curve and finding the depth value corresponding to 80% (Hakanson 1981). Using data from Miller and Szewcyck (1971) to construct the curve, the critical depth in Hamilton Harbour is approximately 1.2 m. This corresponds to an area of roughly 6 km^2. Sediments at depths of less than 1.2 m are susceptible to redistribution by wind. Bathymetric maps show that these sediments occupy a thin zone along the east, west, and north shores. The longest effective fetch occurs with winds from the west, southwest, east, and northeast. Winds from these directions account for 77-84% of the winds in a given month (Atmos. Environ. Ser. 1982). Consequently, the potential for sediment redistribution by wind is high.

The magnitude of wind-induced sediment resuspension depends on the water velocity at the sediment-water interface and sediment characteristics including grain size and cohesiveness (Chesters and Delfino 1978). Fine sediments, for example silt and clay, are more easily resuspended and stay in suspension longer than coarser particles and are more likely to be transported and redistributed to other areas depending on current patterns (Chesters and Delfino 1978). Limited grain size data shows that station 252 has coarser sediments than deeper locations (see Section 2.3.1). If this is true of other nearshore locations, then wind-induced redistribution may not be as quantitatively significant in the harbour as Sly (1977) argues that it is on a lake-wide basis. Chesters and Delfino (1978) estimated that 7000 tonnes/km^2-y were resuspended by wind in the nearshore zone of Lake Ontario between the Niagara River and Rochester. This is probably a minimum of an order of magnitude greater than would occur in the harbour. However, because of limited data there is no way these statements can be evaluated.

The areal distributions of erosion and transportation (a_{E+T}) and deposition (a_A) were also calculated for Cootes Paradise. Substituting $a=1.2$ km^2 (Whillans 1982), $D_{max}=2$ m, and $\bar{D}=0.5$ m (Zarull 1979) resulted in $a_{E+T}=54\%$ and $a_A=46\%$. These calculations may underestimate the area dominated by erosion and transportation processes for a number of reasons. First, Hakanson's (1981) model assumes that wind/wave energy has considerable influence on processes controlling sediment distributions in lakes. This may be true in large lakes, but in lakes smaller than about 1 km^2 this influence (and consequently the a_{E+T} calculation) is limited. The area estimate used for Cootes Paradise is very close to this limit. Second, the model can only be used in single-basin lakes and not in multi-basin lakes where each basin must be treated separately (Hakanson 1981). Better estimates for Cootes Paradise might be obtained if the West Pond and main area of the marsh were treated separately. Lastly, the model is insensitive to changes in length/width ratios and shoreline irregularity. It assumes a standard conical shape for all lakes, with a base area equal to the lake area (a) and the height equal to the maximum depth (D_{max}). The validity of this premise in Cootes Paradise is probably low.

2.4.3 Shipping

Sediment redistribution due to shipping activies in the Great Lakes appears to be important only in shallow channels and harbour areas (Sly 1977; Niimi 1982). Quantitative estimates from other areas are not available, but studies by Sly (1969) and Wilson (1973) provide some information about the potential area affected. Sly (1969), in a study partially designed to test the disturbance of bottom sediments caused by ship movements, found that some disturbance occurred when the total water depth under the ship was less than 6 m. The ship used was a tug with a length of 19.5 m, a beam of 5.0 m, a draft of 1.9 m, and a single propellor (132 x 86 cm with a shaft rotation of 0-500 rpm). When water depths below the propellor axis were less than 2 m extensive sediment stripping occurred to depths of 7-8 cm and for a distance of 6-8 m on either side of the keel. The top 2 cm of sediments were eroded when the water depth was 3.3 m below the propellor axis while with 4.5 m of water, or more, only a slight disturbance of the loose surface materials was noted.

Laboratory studies by Wilson (1973) modelled the effect of a typical seaway freighter (223 m long, 23 m wide, 8.1 m draft) on bottom disturbance. Wilson concluded that the passage of ships with a bottom clearance of 1 m or less will disturb fine sand sediments to a depth exceeding 0.3 m. These figures do not take into account the partial refilling of the scoured area by sediments settling back out after the passage of a ship. Scaling problems with the bed materials led Wilson to suggest that the actual depth of sediment disturbance may be closer to 1 m. Multiple pass tests indicated that the depth of disturbance increased relative to the number of passes. Ten passess produced a scour depth roughly three times that of a single pass. Although bed disturbance may occur in the hull, wake, and propellor zone, disturbances in the wake zone were considered the most important.

Based on these observations Sly (1977) suggested that ships with a bottom clearance of 1-2 m will cause significant disturbance of the bottom sediments to a depth of about 0.5 m and 2-3 times the width of the vessel. Accepting Sly's interpretation suggests that the potential for sediment redistribution by shipping activity in Hamilton Harbour is considerable. The harbour is one of the busiest commercial ports on the Canadian Great Lakes, averaging 689 ships/y for the past five years (Table 14). Barring unforeseen economic events, ship traffic is not expected to change dramatically in the near future (B. Edwards, Ham. Harb. Comm., Hamilton, pers. comm.). Although Table 14 shows that the number of ships using the harbour has declined since the 1950s, total cargo has not changed appreciably because the size of the ships using the port has increased.

Most of the ships using the harbour are lake freighters with a draft of slightly less than 8.1 m. If it is assumed that a bottom clearance of 2 m or less will disturb the sediments, then those in water less than 10 m deep are at risk. This corresponds to an area of approximately 16 km^2 (from the hypsographic curve). Assuming that the depth of disturbance is a uniform 0.5 m, approximately 16 x 10^6 m^3 of sediments may potentially be redistributed by shipping activities. This is an overestimate for a number of reasons. First, it includes sediments from the north and west shores of the harbour that would not normally be disturbed by ships. Second, it is assumed that the entire

Table 14. Number of commercial ships and cargo handled by the Port of Hamilton.

Year	No. of Vessels	Total Cargo ($\times 10^6$ tonnes)
1915	679	
1916	627	
1917	632	
1918	682	
1919	658	
1920	634	
1921-52		
1953	1355	
1954	1296	
1955	1566	
1956	2456	
1957	3690	
1958	2862	
1959	1685	7.92
1960	1615	8.63
1961	1407	8.14
1962	1584	8.58
1963	1625	9.19
1964	1716	9.35
1965	1679	10.29
1966	1662	10.83
1967	1489	10.73
1968	1461	11.96
1969	1367	11.02
1970	1331	12.90
1971	1222	12.44
1972	1029	12.81
1973	1000	13.47
1974	826	11.99
1975	930	14.58
1976	876	13.83
1977	800	12.13
1978	847	13.14
1979	959	14.88
1980	895	14.34
1981	741	10.68
1982	564	8.73
1983	565	10.28
1984	682	12.33

Data provided by B. Edwards, Hamilton Harbour Commissioners, Hamilton, Ontario.

sediment mass would be disturbed, whereas the previously cited studies indicate that only those sediments over a path 2-3 times the width of the vessel will be affectd. Third, it assumes a single pass over the sediments. Most of the vessel movements in the harbour are within well defined approach channels and turning basins. The development of the Eastport facility in the southeast corner of the harbour will concentrate ship traffic in an area that is currently less than 10 m deep. Sediments in this area are heavily contaminated with organic and inorganic compounds and multiple passes through this area may result in the redistribution of sediment from the southeast corner towards the ship canal and deep water areas of the harbour.

2.4.4 Dredging

Although there is very little precise information concerning sediment redistribution by dredging, it is generally thought to be of limited significance quantitatively compared to other mechanisms (Sly 1977). However, if spoils disposal occurs in shallow areas subject to wave and current activity, then widespread dispersal is likely. Given the current dredging and disposal practices in the harbour, it is probable that redistribution is not great, especially since most operations have limited spatial and temporal scope. There is some redistribution in that sediments are taken from the excavation site, but they are effectively isolated in the confined disposal facility and do not contribute to redistribution in other areas of the harbour.

3. DREDGING AND DISPOSAL IN HAMILTON HARBOUR

3.1 Past Dredging and Disposal Activities

Dredging in the Hamilton area dates back to the early 1800s. The first recorded project was the construction of the Desjardins Canal through Cootes Paradise between 1820 and 1837 (Weaver 1982). Soon after its completion a fairly regular maintenance program was established to keep the canal free of sediments brought in by Spencer Creek (Threader et al. 1986). A new channel was dredged between 1851 and 1854 when the present cut through Burlington Heights was constructed (Kerr and Kerr Dec. 14, 1895). The sediment volumes and disposal methods are not known, although the spoils were probably cast into adjacent areas of the marsh.

The first record of dredging in the harbour is the channelization of the Burlington ship canal in 1895 (Kerr and Kerr Dec. 3, 1895). However, no records were kept of the volume of sediment removed or the resulting dimensions of the canal. The Canada Department of Public Works (DPW) has maintained detailed and continuous records documenting annual sediment quantities and the locations of dredging and disposal sites in the harbour since 1898. These data are shown in Table 15 and Fig. 5. It is important to note that they are from operations conducted using public funds only. Some dredging has been done privately, but the data from these projects are not readily available and it is unlikely that the quantities involved are large compared to those shown.

Certain trends are evident in Table 15. Early shoreline restructuring and landfilling along the south shoreline began around 1909 (Ozanian 1957). The construction of wharves made the dredging of channels and berths to meet the

Table 15. Sediment quantities, dredging, and disposal sites in Hamilton Harbour.

Year	Quantity (m^3)	Dredge[a] Site	Dump[a,b] Site
1898	3418	7	
1900-01	-	6,8	
1901-02	1547	8	
1903-04	13962	7,8	
1904-05	37358	7,8	
1905-06	36746	6,7	
1906-07	21102	7,8	
1907-08	40352	7	
1908-09	110453	7	7
1910-11	111778	6,7	7
1911-12	78631	6,7,8	
1913-14	262964	6,7	
1914-15	161614	6	
1915-16	184431	2,6,8	
1916-17	47674	6	
1917-18	32728	2,6,7	
1921-22	4331	2	
1923-24	9024	7,8	
1924-25	5370	2	
1926-27	298	7	
1928-29	269774	6	
1929-30	318427	6,7	
1930-31	576255	6,7	
1931-32	383191	6,7	
1933-34	80095	6,7	
1934-35	91268	6,7	
1938-40	385179	6,7	7
1940-41	-	7	7
1941-42	181977	6	
1947-48	143800	6	
1948-49	6753	7	7
1949-50	110508	6,7,8	1,7
1950-51	24921	2,6,7	1,7
1951-52	109813	5	1,5
1952-53	11584	5	1,5
1953-54	288242	6	1,6
1954-55	2217	2	1
1955-56	154180	5,7	5,7
1956-57	-	7	1,7
1957-59	1503713	7	7
1959-60	1146181	2,3,5,6	1,5,6

Table 15. Continued.

Year	Quantity (m^3)	Dredge[a] Site	Dump[a,b] Site
1960-61	230894	3,7	1,7
1961-62	300469	6,7	6,7
1962-63	17647	7	7
1963-64	77811	7	7
1965-66	94042	5,7	5,7
1966-67	212426	5	5
1967-68	17275	5	5
1968-69	28200	5,7	5,7
1969-70	22002	5,7	5,7
1970-71	22197	5	5
1971-72	35977	5	5
1972-73	917464	5	5
1976-77	127966	5	5
1977-78	19956	7	7
1978-79	87816	6,7	7

Note: Data from Can. Dept. Public Works (1895-1982). Large quantities of sand were removed from area 4 in the harbour for building purposes prior to 1900 (Kerr and Kerr 1860-1898).

a - Refer to Figure 5 for dredging and dump site locations.

b - Spoils disposal is behind berms generally used for dock construction and land reclamation in every area except area 1.

- No data.

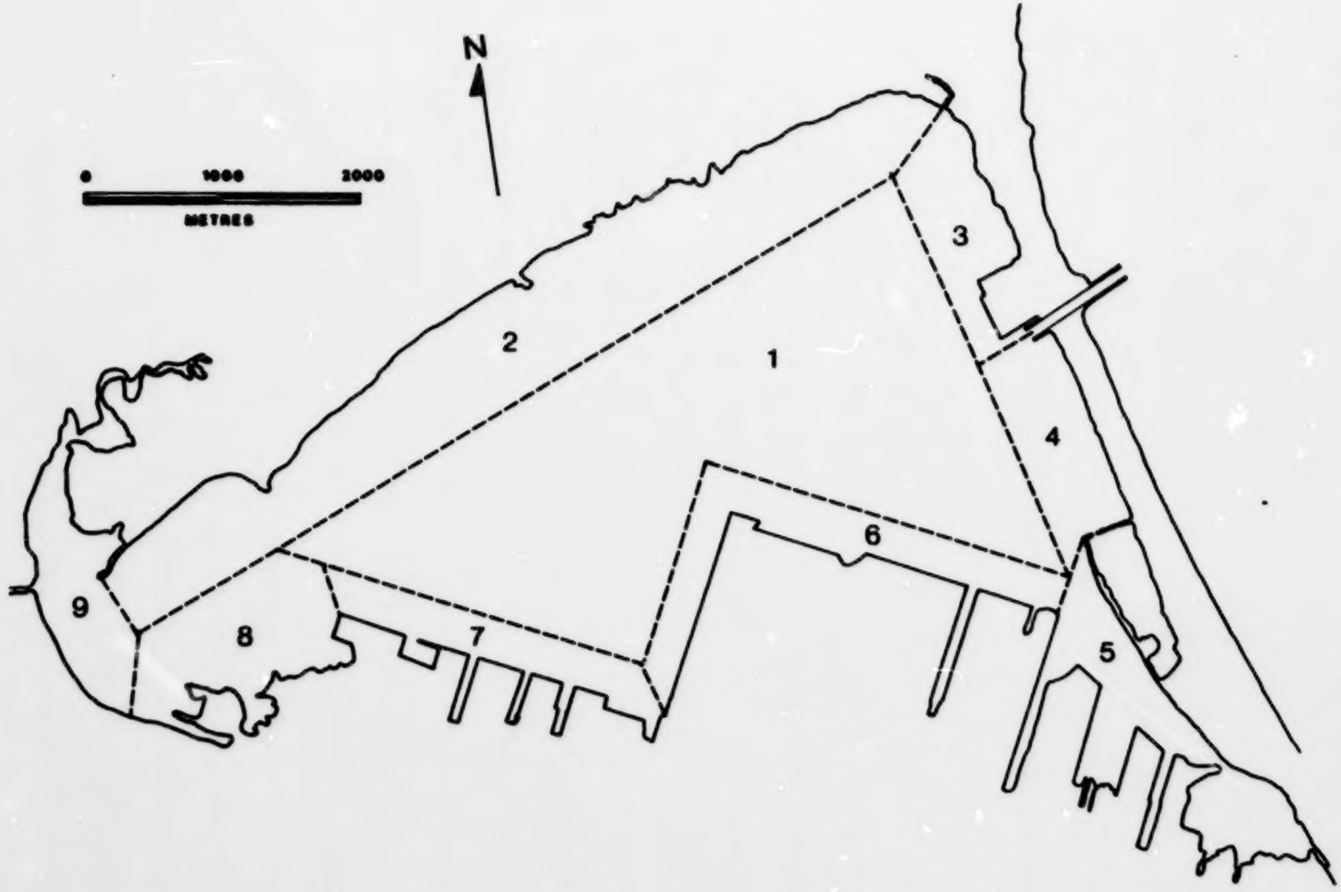


Figure 5. Dredging and disposal site locations in Hamilton Harbour.

navigational requirements of the day necessary and accounts for the increase in dredging after 1909. The majority of dredging in the early decades of this century occurred in the western end of the harbour (areas 7 and 8 in Fig. 5). Occasional projects along the north shore (area 2) were related to maintaining navigational depths around the wharf at LaSalle Park. Although no records of the disposal sites were kept by DPW, Nriagu et al. (1983) indicate that prior to World War II a significant portion of the dredge spoils were dumped into the deeper waters of the harbour (area 1 in Fig. 5).

The present shoreline configuration is the result of extensive wharf construction and land reclamation in the 1950s (west end), the 1960s (east end), and some reconstruction during the 1970s. Stelco and Dofasco also carried out major expansion and land reclamation projects on their properties during the 1950s and 1960s. Most of the wharves were constructed with a perimeter berm of stone or slag and the interior filled with material dredged from the adjacent channels and berths (Grossi 1985). Dredging activity peaked between 1957 and 1960 when the approach channel, turning basin, and slip to the Strathearne Avenue wharves (Piers 23 and 24) were dredged. Although most of the sediments were used in land reclamation during this period, an unspecified quantity was dumped into the deeper waters of the harbour at an unknown location.

Past dredging and disposal activities have influenced habitat changes and the loss of shallow water areas in the harbour. Between 1789 and 1979 approximately 340 ha or 74% of the marsh area in Hamilton Harbour and Cootes Paradise disappeared (Whillans 1982). Most of the loss occurred along the south shore of the harbour. Although some industrial expansion had occurred prior to

1926, most of the marshy areas remained unfilled (Threader et al. 1986). However, by 1982 wharf construction and land reclamation resulted in the filling of 620 ha of marsh and littoral zone with dredge spoils and a 22% reduction in the total area of the harbour relative to 1926 (Threader et al. 1986). Consequently, there is very little high quality fish habitat along the Hamilton waterfront. These losses had a substantial impact on fish community transformations (Whillans 1977, 1979; Holmes and Whillans 1984) and are a key factor limiting fish production.

The disposal of contaminated sediments into the deeper waters of the harbour prior to 1970 has created further environmental problems. These sediments are heavily contaminated with a variety of metals and organic compounds. Core profiles from the deep central basin show elevated levels of Fe, Zn, Mn, Pb, Cu, Ni, and Cd as deep as 52 cm below the sediment surface (Nriagu et al. 1983). Using a Pb-210 dating technique Nriagu et al. were able to show that the depth of the elevated levels roughly corresponded to sediments that were deposited in the period when open-water disposal was prevalent. Contaminant levels in the surficial sediments are slightly less due to the change in disposal strategy in 1970 (see Sect. 3.2.2).

The problem this creates can be interpreted as follows. The heavily contaminated sediments exert about 20-30% of the total oxygen demand in the hypolimnion (Polak and Haffner 1978; Ng 1981) and the harbour is considered an important source of pollutants (most of which are present in the sediments) to western Lake Ontario (Nriagu et al. 1983). A potential solution is to dredge the contaminated sediments from the harbour. However, any disturbance will expose the water column to higher sediment contaminant levels which may lead to increased export into Lake Ontario, add to the oxygen demand, and enhance the possibility that bioaccumulation may occur. Furthermore, the deeper sediments are more heavily contaminated than the surficial sediments, which in turn greatly exceed the guidelines for open-water disposal (see Sect. 3.2.2.1). Containment in confined disposal facilities (CDF) would be necessary, but there is presently no facility with the capacity to contain the deep sediments and little likelihood that any could be built using current construction techniques.

3.2 Current dreding and spoils disposal practices

3.2.1 Current dredging practices

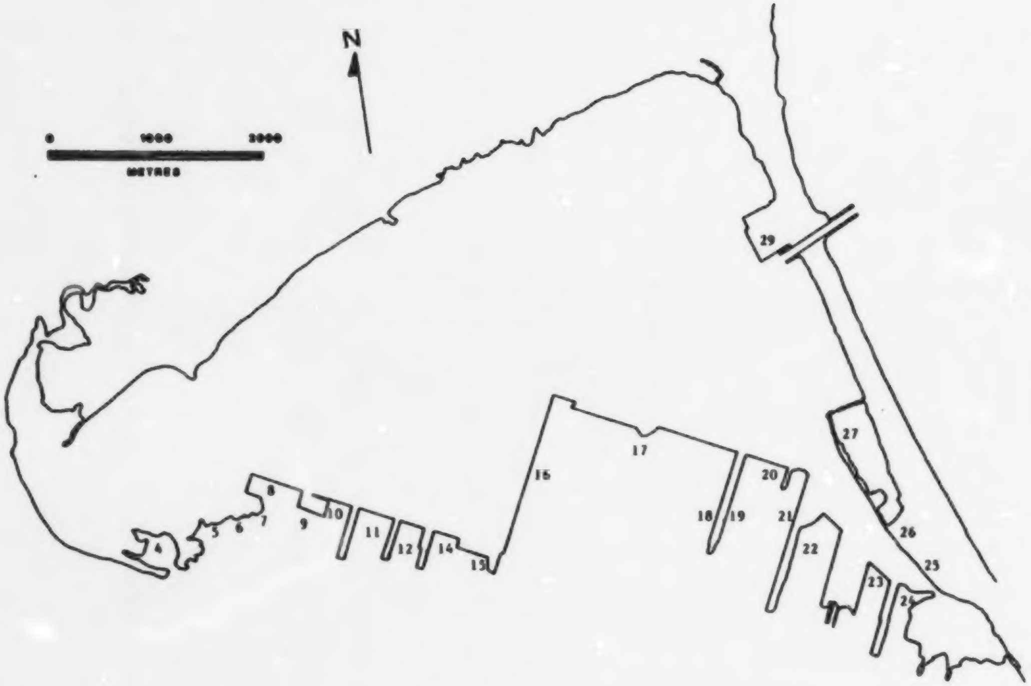
Dredging activity has gradually diminished since the boom in the late 1950s. Currently annual requirements average about 2.5×10^4 m^3/y although maintenance dredging is not an annual event (J. Grossi, Can. Dept. Pubic Works, Toronto, pers. comm.). Although most of the recent activity in the 1980s has been in the approach channel and turning basin to the Dofasco and Strathearne Avenue slips, the data were not included in Table 15 because they were not readily available. Operations east of Stelco generally use hydraulic cutter suction dredges and pump the spoils directly into the disposal site while spoils from west of Stelco are transported by scows and bottom dumped within the confines of the disposal site (Grossi 1985).

Responsibility for maintenance dredging in the harbour rests solely with DPW and Transport Canada. By contrast, capital dredging projects are undertaken by the Hamilton Harbour Commission approximately once every five years.

3.2.2 Current spoils disposal practices

Open-water disposal of harbour sediments has been prohibited since 1970 because of heavy contamination by metals and organic compounds (see Sect. 3.2.2.1). The only operational confined disposal facility (CDF) is the Hamilton Harbour Commissioners' (HHC) East Port complex in the southeast corner of the harbour. This facility, which consists of Piers 25, 26, and 27 (Fig. 6), has a total capacity of 3.0 x 10^6 m^3 and an estimated remaining disposal capacity of 0.5-0.7 x 10^6 m^3 in Pier 27 and a portion of Pier 26 (B. Edwards, pers. comm.; Grossi 1985). During 1977-78 a berm bridging Piers 12 and 13 was constructed and materials dredged from the main channel by Piers 10, 11, and 12 were deposited in the enclosed area. Pier 13 currently forms the eastern side of Pier 12, but it is not shown in Fig. 6 because it is not a separate structure any longer.

The development of the CDF was initiated in 1958 when a portion of the material dredged from the Strathearne Avenue wharves was placed in the Pier 25 location (Grossi 1985). DPW records indicate major berm construction occurred in 1958, 1966, and 1972. The present facility has a series of interior berms dividing it into separate cells. This is designed to improve slurry deposition from hydraulic equipment and the containment of contaminants through enhanced settlement of the fine sediments (Grossi 1985). Table 16 shows the effect of the cells on effluent quality. These data show that water quality in the last cell (4) is comparable to ambient water quality in the harbour. When scow dumping is necessary, the north berm along Pier 27 (Fig. 6) is temporarily cut open to allow passage into a reservoir cell within cell 4 (Grossi 1985). Once this cell



LEGEND

Pier	Facilities
4	Park
5	Royal Hamilton Yacht Club, HHC launching ramp, Leander Boat Club
6,7	HHC Harbour Police, HHC Marine dockyard
8	HHC Catharine St. terminal, Laidlaw Transport Ltd., Navy League of Canada, park
9	HMCS Star
10	HHC Wellington St. terminal, McKeil Work Boats Ltd., Bermingham Construction Ltd., Mission to Seamen, Hendrie & Co., Amstel Breweries Ltd.
11	Gillies-Guy, Red-D Mix Concrete Co., Pioneer Service Station, Ultramar Canada Inc., Canada Transport Ltd., Can. Veg. Oil Processing, Ship Repairs and Supplies Ltd., Best-Mark Enterprises, Marko Welding Service
12	Seaway Terminals, Shell Canada Ltd., Myers Fuels Ltd., United Co-operatives of Ontario, Wakeham & Sons
14	Seaway Terminals, Canadian Dredge and Dock, Baldasare and MacGregor Ltd., Weights and Measures (Govt. Can.)
15	International Harvester
16,17,18	Steel Co. Canada (Stelco)
19,20,21	Dominion Foundries and Steel (Dofasco)
22	Stelco, National Steel Car
23	Usarco Ltd., A.J. Frank & Son, Construction Aggregates Ltd., Firestone Tire and Rubber, Canada West Indies Molasses Co., Art Gray Diving, Sceptre Dredging
24	Kingsway Transport, Canada Steamship Lines Inc., Frontenac Chemical Waste Service Ltd.
25,26	HHC East Port Industrial Park
27	Current confined dredge spoils disposal area
29	Canada Centre for Inland Waters (CCIW)

Figure 6. Piers and facilities in Hamilton Harbour. From Ham. Harb. Comm. (1983).

Table 16. The effect of interior cells on effluent quality in the Hamilton Harbour confined disposal facility.

Parameter	Cell[a] 2	3	4	Ambient[b]
PCB (ug/L)	40	3	0.5	-
Oil & Grease	360	10	5	-
As	5.4	0.003	0.001	0.001
Cd	0.331	0.002	0.002	<0.0002
Cr	93.5	0.11	0.02	-
Cu	16.4	0.015	0.05	0.007
Pb	16.3	0.03	0.03	0.004
Hg (ug/L)	0.16	0.1	0.1	0.04
Ni	4.96	0.068	0.039	0.005
Zn	96.5	0.135	0.04	0.02
Suspended Solids	26316	64	14	-

Note: All figures are in mg/L unless otherwise indicated.
a - Mean concentrations in cells taken from Grossi (1985).
b - Data are taken from OME (1985) and are 1982 median values at station 4 based on 36 samples for each metal, except 18 for Hg and As.
- No data available.

is filled the material will be rehandled into other cells for final disposal. The interior berms also permit staged development of the site once the disposal function is completed. The Hamilton Harbour Commission is presently developing Pier 25 for industrial purposes.

3.2.2.1 Administration and regulation of spoils disposal

The Hamilton Harbour Commission was created by a federal Act of Parliament in 1912. The Harbour Commissioners have the authority to regulate the "excavation, removal or deposit of material or any other action that is likely to affect in any way the docks, piers, wharfs or channels of the harbour or adjacent lands" (RSC 1970, S.13(1)(c)). Theoretically, they have jurisdiction over all dredging and disposal operations in the harbour. Practically, however, their involvement is limited to the administration of capital dredging projects and the operation of the CDF.

Although the harbour is a multiple use facility, the principal of "navigable servitude" has been used by the courts to interpret the priority amongst uses (Sproule-Jones 1985). Where shipping and navigation interests conflict with other uses of the harbour, then shipping and navigation have priority (Sproule-Jones 1985). The authority to dredge comes from the Navigable Waters Protection Act administered by Transport Canada (Castrilli 1977). The Act protects waters for navigation through provisions

> that no work may be built or placed in, upon, over, under, through or across any navigable water unless the work and the site and the plans have been approved by the Minister, upon such terms and conditions as he deems fit (RSC 1970, S. 5(1)(a)).

If the work does not interfer with navigation in a substantial way then Ministerial approval is not normally required (Castrilli 1977).

Prior to 1982 maintenance dredging in the harbour was administered by the Canada Department of Public Works (DPW). Public Works also drew up the project specifications, tested the sediments, tendered the contracts to private companies to perform the work, and supervised the on-site activities (J. Grossi, pers. comm.). The Ports and Harbours Branch of Transport Canada assumed the administrative and funding roles for commercial harbours, including Commission harbours, in 1982 (D. Mattingly, Ports Harb. Br., Trans. Can., Toronto, pers. comm.). These responsibilities were recently transferred to the Coast Guard Branch which shares them with the Harbour Commissions in Commission harbours. Current policy is that Transport Canada will administer and fund the dredging of primary channels while the Harbour Commission assumes these responsibilities for secondary channels and berths. The exact boundaries of these channels were defined by mutual agreement between the two agencies. The DPW has an agent-client relationship with both wherein DPW does the engineering and supervision for their clients, who administer and pay for the project.

Since dredging in the harbour is a federal responsibility all proposals are normally subjected to the environmental assessment and review process (EARP) by the Enviromental Protection Service (EPS) of Environment Canada (Castrilli 1977). The suitability of dredged sediments for open-water disposal is assessed using provincial Ministry of the Environment (OME) guidelines established in 1976 and revised in 1978. These criteria are shown in Table 17 along with guidelines used by the United States EPA, Region V. It is the policy of OME to

Table 17. Summary of the guidelines used to assess the acceptability of sediments for open-water disposal in the Great Lakes.

Parameter	OME[a]	EPA[a,b]	
	(1976)	NP	HP
Volatile Solids[c]	6%	<5%	>8%
COD	5%	<40000	>80000
TKN	0.2%	<1000	>2000
Total P[d]	0.1%	<420	>650
Oil and Grease	0.15%	<1000	>2000
PCBs	0.05	≥10[e]	
Hg	0.3	≥1[e]	
Pb	50	<40	>60
Zn	100	<90	>200
Fe	10000	<17000	>25000
Cr	25	<25	>75
Cu	25	<25	>50
As	8.0	<3	>8
Cd	1.0	f	>6
Cyanide	0.1	<0.10	>0.25
Ammonia	100	<75	>200
Ni	25	<20	>50
Mn	-	<300	>500
Ba	-	<20	>60

Note: All figures are in mg/kg unless otherwise indicated.

a - From Dredging Subcommittee (1982). Analysis must be on a dry weight basis.

b - NP-nonpolluted; HP-heavily polluted. Values falling between are in the moderately polluted category which is not shown.

c - Loss on ignition at 600°C.

d - As P.

e - If these values are exceeded, the sediments are considered polluted and unacceptable for open-water disposal even if other parameters are well below their established criteria.

f - Limits not yet established in non- and moderately polluted categories.

allow open-water disposal of dredge spoils only when the contaminant contents of the spoils meet or exceed the existing guidelines and provided that other water uses in the area are unaffected (Dredging Subcommittee 1982). There are no existing federal or municipal regulations that affect spoils disposal in the harbour.

The present EPA and OME guidelines are based solely on bulk chemical characterization of sediment. The adverse effects of these chemicals in combination are freqeuntly unknown and the Dredging Subcommittee of the IJC is urging agencies to incorporate biological responses (bioassays) into revised guidelines.

Recent sediment contaminant data for Hamilton Harbour are compared with the OME disposal guidelines in Table 18. These data show that harbour sediments are grossly contaminated by all analyzed parameters except Hg, which is equal to its guideline, and Mn, for which no criterion has been established. These data are means for all sediment sampling locations in the harbour (Fig. 3) including those along the north shore and in the west end. These stations are considered relatively unpolluted compared to stations along the south shore and in the deep central basin.

Spoils exceeding the OME criteria in any parameters may be considered unacceptable for open-water disposal on the basis of water quality, water use or public health reasons (Dredging Subcommittee 1982). Such material must be confined either within a dyked area or on land, in a manner that prevents the entry of pollutants into a watercourse (Dredging Subcommittee 1982). Although the degree of confinement may vary depending on the pollutant, for some such as PCBs and Hg, no escape should occur.

Table 18. Comparison of 1980 sediment analysis data from Hamilton Harbour with OME open-water disposal guidelines.

Parameter	Oct 1980[a]		OME Guideline
	Mean	SE	
Volatile Solids (%)	9.8*	2.2	6
TKN (%)	0.35*	0.04	0.2
Total P (%)	0.32*	0.05	0.1
Hg	0.3	0.04	0.3
Pb	310.0*	45.0	50
Zn	2700.0*	380.0	100
Fe	68000.0*	12000.0	10000
Cr	204.0*	38.0	25
Cu	130.0*	18.0	25
As	19.9*[b]	11.3[b]	8.0
Cd	5.7*	0.8	1.0
Ni	44.0*	4.0	25
Mn	1900.0	224.0	-

Note: All figures in ug/g (dry weight basis) unless otherwise indicated. All data from OME (1985).

a - Mean of top 3 cm of surficial sediments from 20 locations sampled in October 1980.

b - Top 3 cm of surficial sediments from 55 locations sampled in October 1984 by (Portt et al. 1986).

- No data available.

* - MOE open-water disposal criterion exceeded.

The 1976 OME guidelines have limited applicability as indicators of potential hazard to aquatic ecosystems. First, the criteria for open-water disposal are based on bulk sediment analysis. Changes in substrate type may change contaminant concentrations because most contaminants are associated with the finer size fractions (Dredging Subcommittee 1982). Second, the individual values were empirically derived and do not attempt to relate contaminant concentrations to specific adverse impacts on aquatic biota (Boyd and Griffiths 1985). Third, the guidelines provide information on total concentrations only. Correlating total concentration with biological availability and adverse effects under natural environmental conditions has not yet been satisfactorily accomplished (IWGACPDA 1975). Wide variations among seasons and in natural background levels as well as variations in geochemical and biochemical reactions under aerobic and anoxic conditions have hampered such attempts. To overcome some of these problems, the EPA developed the ranges for each parameter shown in Table 17.

Other problems are also associated with the guidelines. Most of the individual parameter levels are derived from laboratory studies. It is extremely difficult to relate data obtained under controlled conditions to widely variable field conditions. For example, contaminants consistently released in the elutriate test developed by the U.S. Army Corps of Engineers and actual disposal operations included Mn and ammonia (Lee 1976; Wright 1978). Phosphorus is frequently released during disposal operations and less frequently during the elutriate test while TOC and As releases are less frequently observed in field testing but are commonly noted during elutriate assays (Sly 1977;

Wright 1978). One further problem is the increasing evidence that mixtures of metals may have additive, synergistic or antagonistic effects on the bioavailability and toxicity of the individual metals involved (Wong et al. 1978). Selenium, for example, can reduce the accumulation of Hg in aquatic food webs (Rudd et al. 1980).

Both federal and provincial officials began to revise the disposal guidelines in the early 1980s. New joint federal-provincial guidelines for open-water disposal are expected to be published within a year (I. Orchard, Env. Prot. Ser., Env. Can., Toronto, pers. comm.). Although some of the chemical criteria for disposal may change, these changes are not expected to be large. The main difference will be the emphasis on priority chemicals such as PCBs and Hg. New regulations are expected to contain some measure of ecosystem response determined from screening bioassays. Suitable test procedures are presently being developed by U.S. and Canadian agencies (Dredging Subcommittee 1982).

3.2.2.2 Waterlots

The major waterlots and ownership are shown in Fig. 7. The numbers in this figure refer to pier numbers assigned by the Hamilton Harbour Commission. The waterlot boundaries shown must be considered provisional since new surveys are needed to establish exact boundaries with certainty. This is particularly true of the CNR waterlot, the waterlot in front of Piers 16 and 17 of Stelco's main Hamilton facility, and the waterlot in front of Dofasco (Pier 20).

Waterlot ownership has been fairly stable. All properties on the harbour including Cootes Paradise were deeded to the City of Hamilton in 1846 (Ham.

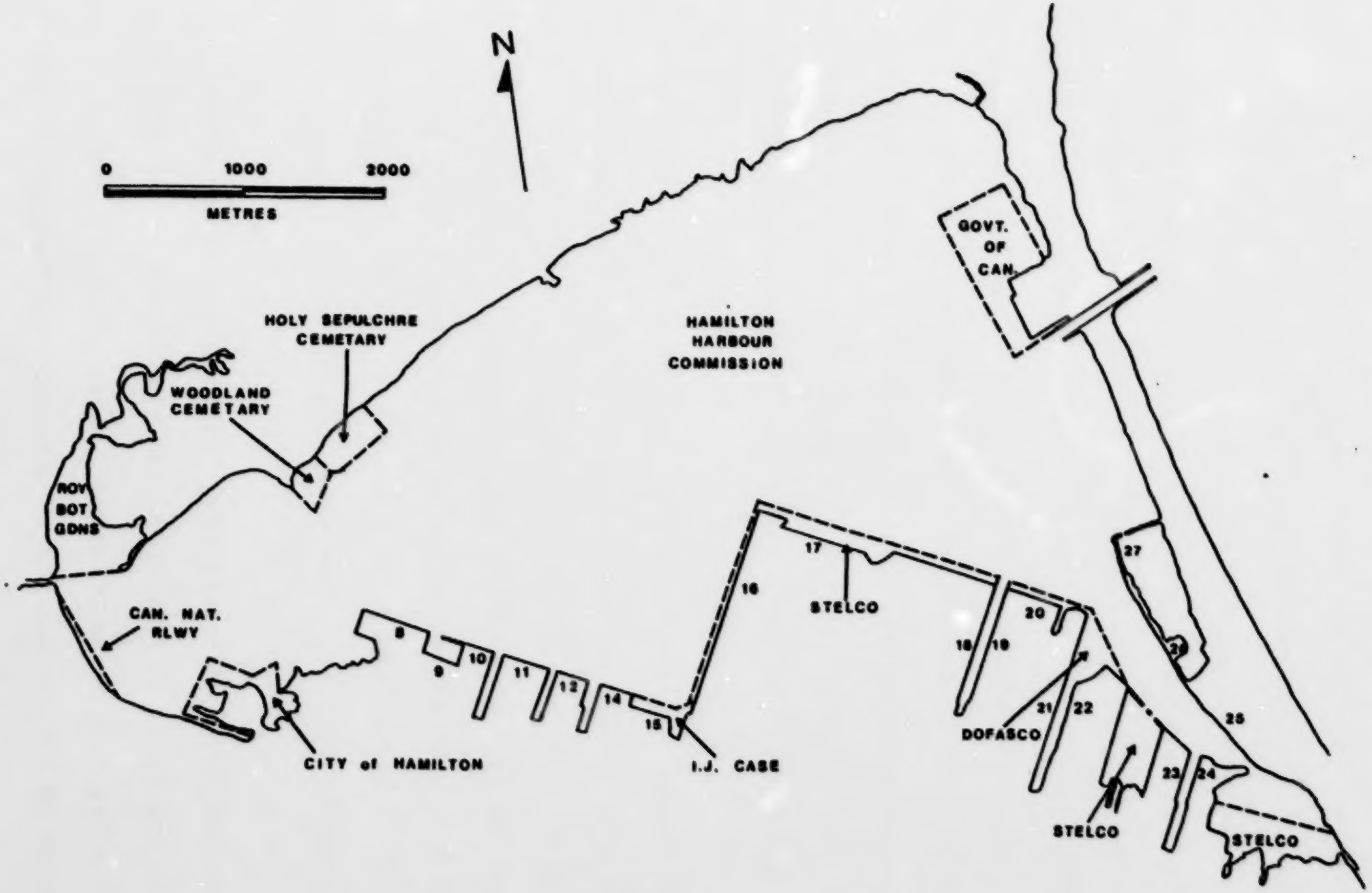


Figure 7. Major waterlots and owners in Hamilton Harbour.

Harb. Comm. 1983). Cootes Paradise and the waterlot northwest of Carroll's Point were given to the Royal Botanical Gardens (RBG) in 1941. In 1948 the bed of the harbour was deeded by the City to the Hamilton Harbour Commission (Ham. Harb. Comm. 1983). The Harbour Commission currently owns all the waterlots in the harbour except for some small privately owned parcels along the north shore and those shown in Fig. 7. Both Stelco (with three lots) and the federal government (with the bed around the Canada Centre for Inland Waters) also hold relatively large areas of the harbour bed. The City of Hamilton recently obtained title to the waterlot surrounding the former Lax property at the western end of the harbour. The cemetary waterlots extend approximately 150 m offshore and apparently were bought to prevent industrial development along the shoreline (B. Edwards, pers. comm.).

Filling of waterlots in the future could have significant implications for fisheries rehabilitation in the harbour. The only waterlots being filled presently, are the Pier 26 and 27 lots belonging to the Hamilton Harbour Commission. Filling is expected to be complete around the year 2000 after which the 50 ha area will be developed by the Hamilton Harbour Commission for marine and/or industrial purposes (Ham. Harb. Comm. 1983). The Harbour Commission has no existing plans to fill any other waterlots in the forseeable future (B. Edwards, pers. comm.).

Plans for the other waterlots vary. Stelco recently constructed a berm joining Piers 22 and 23 and is filling the waterlot behind it and will probably extend Pier 17 outward in the near future. About 10 ha were landfilled along the eastern margin of the harbour due to arterial road realignment associated with the twinning of the Burlington Skyway bridge (Ont. Min. Trans. Comm. 1982). Waterlots were also filled in the late 1960s during the construction of CCIW. The CNR has been filling its waterlot for years to prevent slumping along its right-of-way. Future plans are uncertain because there is some doubt as to the extent of the remaining unfilled portion of the waterlot.

The plans of the remaining waterlot owners are not known. Some filling for shoreline protection and to prevent slumping along the north shore are possibilities. The quantity of fill and the total area involved are not likely to be large. No filling is expected in the Royal Botanical Gardens or City of Hamilton waterlots.

3.3 Future plans for dredging and spoils disposal

Maintenance dredging in the future is expected to involve similar sediment quantities, frequencies, and locations, given the present sediment loading rates and the principal of "navigable servitude". Some capital dredging is also expected to be carried out by the Hamilton Harbour Commission around Pier 11 and along the west face of Piers 25, 26, and 27 to bring them to Seaway depth of 8.2 m below datum (B. Edwards, pers. comm.). The sediment volumes from the capital dredging are not expected to be significant compared to the quantities produced by future maintenance projects.

Future spoils disposal will continue to require CDFs because of heavy pollutant loads. The East Port CDF is expected to last until about 2000 if present filling rates are maintained. Requirements for additional containment capacity after this date will be problematical because there are no other readily apparent sites. One option is to extend the present facility north. This may be difficult because water depths rapidly increase to 15-20 m. New berm construction techniques would be required since the width of berms built using existing methods would greatly reduce disposal capacity (B. Edwards, pers. comm.). There is no indication that serious planning has gone into future spoils disposal scenarios after 2000. This should be initiated to ensure that environmentally safe methods are available when they are needed.

Water and sediment quality data from Windermere Basin indicate some degradation has occurred. Concern has been expressed that the filling of the basin with sediments from Redhill Creek and the Hamilton STP may result in higher contaminant loadings to the harbour proper. Envirosearch (1985) evaluated the alternatives available to isolate and retain the contaminated sediments in Windermere Basin from the aquatic environment and concluded that on-site disposal was the most viable procedure. Although there are a number of options, all involve partial dredging and the creation of a containment area within a portion of the basin. The Hamilton Harbour Commission has indicated that it does not have the capacity in its CDF. In addition, they are reluctant to accept reclaimed land next to the East Port complex. Stelco, the other waterlot owner, may not have similar objections. A final decision to resolve the problem will be made in the near future by the Windermere Basin Advisory Committee and the OME.

4. IMPACTS OF DREDGING AND DISPOSAL ON HAMILTON HARBOUR FISHERIES

4.1 Effects on primary and secondary productivity

There is evidence in the literature that dredging and disposal may alter plankton productivity (Munawar 1982; Munawar et al. 1983). Plankton, zooplankton in particular, are an important component in the diet of most larval fish and the adults of species such as herring, alewife, and gizzard shad. Declines in zooplankton productivity will have a detrimental effect on obligate planktivores such as alewifes, while opportunistic species such as trout and yellow perch will readily switch to other prey if zooplankton are not available. Phytoplankton are less often consumed because the majority are too small to be seen by planktivorous fish, which actively search for and select their prey. Nevertheless, changes in phytoplankton productivity will have an indirect effect on fish productivity because many zooplankton depend on phytoplankton as their food source.

The phytoplankton community in Hamilton Harbour consists of species typical of highly eutrophic lakes (Piccinin 1977). While the majority are chlorophytes and cryptophytes, small numbers of diatoms and cyanophytes are also present. Representative species of each group respectively include: Tetrastrum sp. and Chlamydomonas sp.; Rhodomonas minuta and Cryptomonas sp.; Synedra ulna and Asterionella formosa; and Oscillatoria sp. and Anabaena sp. (OME 1978, 1985). These species display broad seasonal cycles that are similar from year to year (Harris and Piccinin 1980). Chlorophytes dominate during the spring and late summer while cryptophytes are the dominant group during the early summer months. Diatoms and cyanophytes are found throughout the summer with peak diatom densities occuring in July and lesser peaks in May and August.

Nutrient loadings to the harbour are large and continuous (10.5 g $P/m^2/y$ and 320 g $N/m^2/y$) and total phosphorus concentration in the water column averages between 60-95 ug/L with monthly means as high as 279 ug/L at certain stations (Harris et al. 1980a). Data from Harris et al. (1980a) does not show any evidence that phytoplankton are limited by nutrient availability, yet summer algal biomass (as chlorophyll a) ranges from 20 to 70 mg/m^3. These values are well below those predicted by the loadings of total P or total P concentrations in the water column. Further work by Harris et al. (1980b) indicates that physical variables including light, temperature, and vertical mixing are important factors controlling phytoplankton productivity in the harbour. Lake-harbour exchanges through the ship canal are also an important controlling factor.

The euphotic zone normally ranges in depth from 2 to 4 m during the ice free period (Harris et al. 1980a). High turbidity, absorption by phytoplankton, and most importantly, absorption by dissolved organic compounds in the water column are responsible for the shallowness of this zone. Roughly 80% of the phytoplankton biomass is produced in the top 2 m of water (Harris 1976; Piccinin 1977).

The thermal structure of the water column strongly influences population composition, size, and frequently the life cycles of particular species (Haffner et al. 1980). The harbour is dimictic, but stratification during the summer and the depth of the thermocline (2-12 m) are highly unstable (Harris et al. 1980a). These perturbations are created by internal oscillations of the thermocline from seiche and internal wave motions and the larger scale oscillations of Lake Ontario that are superimposed on the internal motions by exchanges through the ship canal (Harris et al. 1980a).

The large and frequent changes in the depth of the thermocline and the shallow euphotic zone are the major factors limiting algal productivity in the harbour (OME 1985). Data from Harris et al. (1980b) shows that during periods of physical stability, phytoplankton productivity increases such that significant self-shading occurs. These periods are often characterized by populations of Cryptomonas sp. (Haffner et al. 1980).

Most of the current dredging operations occur in certain channels and turning areas of the southeast portion of the harbour on an irregular basis. These operations may have local impacts, but it is unlikely that they could be detected above the normal variability in phytoplankton populations. The influence of harbour-wide physical variability on phytoplankton populations probably overshadows all other factors.

Similarly, the available evidence suggests that temporary increases in turbidity and nutrient concentrations are not likely to have much effect on phytoplankton productivity. Most populations are already well adapted to the high turbidity levels in the harbour and it is somewhat uncertain whether dredging would materially add to these levels. Additionally, Haffner et al. (1980) have pointed out that some phytoplankton in the harbour may avoid light-limited environments by means of resting stages or heterotrophy. Finally, biomass data shows that growth is not limited by nutrients, therefore, increases in nutrient loadings due to dredging should have no immediate effect on algal biomass.

The effects of contaminant releases on phytoplankton productivity in the harbour are not clearly known. Approximately 50% of the species that have been identified are from the <20 um size fraction (Table 19). This size fraction is more sensitive than the >20 um fraction to a variety of contaminants (Munawar 1982; Munawar et al. 1983, 1985), including most of the heavy metals found in the surficial sediments of Hamilton Harbour (see Tables 11 and 12). Given their demonstrated sensitivity and the long history of contamination, it is unlikely that the <20 um fraction comprises a major portion of the phytoplankton biomass in Hamilton Harbour. If phytoplankton biomass in the harbour is dominated by larger species, then the effects of current dredging and disposal activities may be partially mitigated, particularly since contaminant releases from the sediments tend to be localized and site-specific in nature (see Section 1.2.4).

Zooplankton populations in the harbour are dominated by rotifers, with smaller populations of cladocerans and copepods (Harris 1976; Piccinin 1977). These species also display a seasonal cycle with the peak biomass in the spring being composed almost entirely of Keratella sp. (Harris 1976). During the early summer, zooplankton biomass declines and remains at low levels until the fall when another increase occurs. The fall increase is particularly noticeable in deep areas probably because hypolimnetic aeration associated with the fall overturn opens up new habitat.

The dominant zooplankton taxa in the harbour are mostly pelagic forms; with natural populations being governed by the quantity of available food (i.e phytoplankton) (Wetzel 1975). Work by Harris (1976) indicates that temporal fluctuations in zooplankton productivity are correlated with fluctuations in

Table 19. Species and size composition of the natural phytoplankton assemblage in Hamilton Harbour, 1975-1977.[a]

<20 um Ultraplankton	>20 um Microplankton/Netplankton
Chlamydomonas sphagnicola	*Pediastrum duplex*
Oocystis Borgei	*Ankistrodesmus falcatus*
Scenedesmus quadricauda	*Anabaena flos-aquae*
Scenedesmus acuminatus[b]	*Staurastrum paradoxum*
Scenedesmus denticulatus[b]	*Asterionella formosa*
Scenedesmus bijuga	*Fragilaria crotonensis*
Selenastrum minutum	*Cryptomonas ovata*
Tetrastrum staurogeniaforme	*Cryptomonas rostrata*
Dictyosphaerium pulchellum	*Diatoma elongatum*
Rhodomonas minuta	*Synedra acus*
Stephanodiscus astraea	*Melosira granulata*
Stephanodiscus Hantzschii	*Closteriopsis longissima*
Cyclotella meneghiniana[b]	*Pandorina morum*[b]
Merismopedia punctata	*Eudorina elegans*[b]
Chroococcus minutus[b]	*Actinastrum Hantzchii*[b]
Tetradesmus wisconsinensis[b]	*Coelastrum microporum*[b]
Crucigenia tetrapedia[b]	*Micractinium pusillum*[b]

Note: Composite species list from Harris and Piccinin (1980) based on sampling at stations 258 and 270 (see Fig. 3) during the period 1975 to 1977. Size classifications are based on information in Munawar (1982), and Munawar et al. (1983, 1985) unless otherwise indicated.

a - The following taxa were found in the harbour but were not placed in either category because of insufficient size and/or species information: *Echinosphaerella limnetica*, *Treubaria setigerum*, *Lagerheimia longiseta*, *Quadrigula lacustris*, *Golenkinia paucispina*, *Cosmarium circulare*, *Closterium acerosum*, *Franciea Droescheri*, *Trachelomonas* sp., *Euglena* sp., *Phacus* sp., *Peridinium* sp., *Kirchneriella* sp., *Tetraedron* sp., *Oscillatoria* sp., and *Mougeotia* sp.

b - Size classifications based on size information in Findlay and Kling (1979). Colonial species are classified using colony size *not* individual cell size.

phytoplankton biomass and the thermal structure of the water column. Since zooplankton grazing rates and prey selection in the harbour have never been determined, it is presumed that zooplankton productivity is controlled to a large degree by the physical variability of the water column. As with phytoplankton, it is doubtful whether dredging operations have any great effect on zooplankton productivity, especially since the productive deep areas are far removed from dredging and disposal sites in the harbour.

There is very little recent information about the composition or distribution of the phytoplankton or zooplankton communities in the harbour. The pelagic habits of most of the known plankton species suggest that dredging may have minimal effects on plankton productivity. This is not expected to significantly affect the fish community because the changes are restricted to a region of the harbour supporting few fish and only a small number of resident species are obligate planktivores.

4.2 Effects on macrophyte and benthic productivity

Macrophytes are important indicators of water quality and are limiting factors controlling the growth, production, and distribution of fish in lakes (Jaworski and Raphael 1979). The removal and disposal of dredged materials in the littoral zone has the potential to create significant changes in macrophyte communities. Frequent maintenance dredging and confined disposal in the littoral zone will permanently reduce macrophyte productivity. In contrast, capital dredging sites may be recolonized rather quickly although some changes in species composition and productivity may occur. Changes in species distribution and/or diversity will cause fairly rapid and compensatory changes in the fish community.

There is little precise information describing the current distribution and abundance of macrophytes in Hamilton Harbour. The Royal Botanical Gardens conducted detailed surveys in 1984 and 1985 (L. Simser, Roy. Bot. Gdns., Hamilton, pers. comm.) and found that a narrow, nearly continuous zone of macrophytes extends along the north and west shorelines in water less than 2.5 m deep. Species diversity and density were greatest in the area around Carroll's Point, but declined rapidly to the east and west. The Hamilton waterfront between Pier 8 and Windermere Basin was devoid of all macrophyte beds. Although this information is rudimentary, it suggests that the impact of current dredging and disposal activities on the macrophyte community in the harbour is probably minimal.

The distribution and abundance of macrophytes in Cootes Paradise have been well documented. The area of submergent and emergent vegetation has declined approximately 50% since the 1950s (Threader et al. 1986). This may be due to a number of factors including carp (Simser 1982), water level fluctuations (Whillans 1977, 1979), the destabilization and resuspension of existing sediments due to wind and wave action (at least 54% of the sediment surface in Cootes Paradise is subject to erosion and transportation processes from wind and wave activity; see Sect. 2.4.2), suffocation by settling sediments (e.g. Lamoureux 1958), and finally, filling due to sediment inputs from land erosion, especially in the western end from Spencer Creek (Semkin et al. 1977). Further efforts by the Departments of Fisheries and Oceans and Environment at the Canada Centre for Inland Waters are underway to determine more clearly the roles of fluctuating water levels, sediment inputs, and disturbances by carp in regulating the distribution and abundance of aquatic macrophytes in Cootes Paradise.

The distribution and diversity of benthic invertebrates are routinely used to assess water and sediment quality in lakes. Natural populations generally exhibit two maxima of abundance and biomass with depth, one in the littoral zone and the other in the profundal zone (Wetzel 1975). Although food supply seems to be the dominant factor controlling productivity, fish predation may also have major effects (Wetzel 1975). However, changes in benthic productivity from other causes will also affect fish productivity.

The macroinvertebrate community in Hamilton Harbour was first described by Johnson and Matheson (1968). These authors sampled 46 locations in 1964 and found that nine oligochaete species, particularly Limnodrilus hoffmeisteri and Tubifex tubifex, were the dominant invertebrates in the harbour. The only other animals identified were four chironomid genera. Densities of L. hoffmeisteri and T. tubifex ranged from 1000 to 10,000 animals/m^2 in profundal sediments rich in organic matter. A more diverse but less abundant oligochaete community composed of five Limnodrilus species other than L. hoffmeisteri was found in the less organically enriched sublittoral sediments of the west end, north shore, and near the Burlington Canal. Low numbers of chironomids were also found along the north shore and near the Burlington Canal. No macroinvertebrate community occurred in about 2 km^2 of sediments centred on the steel mills apparently because the iron oxide content in these sediments exceeded 25%.

The status of the macroinvertebrate community improved considerably by

1984. Portt et al. (1986) resampled the 1964 sites and found that oligochaetes, particularly L. hoffmeisteri and Peloscolex multisetosus, were the dominant invertebrate taxa in the harbour. Altogether they identified ten species of oligochaetes, eight chironomid genera, and nine species of sphaeriid clams. In addition, species abundances throughout the harbour were higher compared to 1964 and the toxic zone near the steel mills was reduced to a small area in the Ottawa St. slip between Piers 18 and 19 (see Fig. 6). The presence of P. multisetosus (Brinkhurst 1970) and the sphaeriids (Pennak 1978) indicates that there is less organic pollution in 1984 compared to 1964 as a result of improved municipal and industrial waste treatment practices.

Current dredging and disposal operations are not likely to signficantly alter macroinvertebrate productivity in the harbour, regardless of the recent improvements noted above. Johnson and Matheson (1968) found that oligochaete densities along the Hamilton waterfront and in Windermere Basin ranged from 0 to about 4000 animals/m^2. These figures are relatively low compared to the harbour-wide range off 70 to 22,600 animals/m^2. Similarly, although recent oligochaete densities are higher (perhaps 1000-10,000 animals/m^2; Portt et al. 1986), they remain low compared to other areas of the harbour. The low abundances combined with the limited area of sediments affected by dredging and spoils disposal operations suggests that macroinvertebrate productivity will not be seriously affected. It is important to note that historically the coldwater fishery in the harbour (especially lake whitefish) may have relied on benthic invertebrates, particularly Pontoporeia affinis, as a major food source. There are currently no P. affinis in the harbour possibly as a result of low oxygen levels or toxic sediments and it is doubtful if the coldwater fishery could be rehabilitated in the absence of a more diverse benthic community.

4.3 Effects on fisheries

The fish community in Hamilton Harbour has undergone numerous changes from early settlement to the present. The harbour once supported a viable coldwater and a productive warmwater fishery. Lake trout, whitefish, and herring were permanent residents of the deep central basin while large populations of largemouth bass, smallmouth bass, northern pike, and other warmwater species inhabited the extensive littoral area along the north and south shores (Whillans 1977, 1979; Holmes and Whillans 1984). These fisheries gradually responded to a variety of stresses including exploitation, habitat destruction, and most importantly, deteriorating water quality. By the early 1900s the coldwater fishery was virtually eliminated and the warmwater fishery was showing strong signs of deterioration. Several stress periods resulting in fish community transformations were described and interpreted by Whillans (1977, 1979). The first occurred in the late 1800s and the last in 1961. Currently carp, carp-goldfish hybrid, white sucker, white perch, and brown bullhead are the dominant taxa in the harbour.

At least 18 of the cultural stresses outlined by Francis et al. (1979) are known to have been active in Hamilton Harbour, but from a fisheries perspective only two are of potential concern in this report: dredging and shoreline restructuring (spoils disposal). These two primary stresses are associated with four indirect stresses: turbidity, oxygen depletion, nutrients, and contaminants. The indirect stresses tend to have temporary effects which are intensified by the direct (primary) stresses (Acres 1983). The persistent effects of the primary stresses tend to be intensified at the disposal site, but mitigated elsewhere (Acres 1983).

There is good evidence linking reductions in benthic invertebrate and macrophyte productivity with past dredging and disposal activities in the harbour. Between 1789 and 1979 approximately 3.4 km^2 or 74% of the marsh area in the Hamilton Harbour-Cootes Paradise ecosystem disappeared (Whillans 1982). Most of this productivity was lost from the south shore of the harbour largely due to spoils disposal in the shallow littoral zone. Although the excavation process may cause some disruption, the productivity effects are quantitatively insignificant compared to those accruing from spoils disposal. It is assumed that the high autochthonous productivity and habitat diversity of this area supported relatively high benthic productivity. Some benthic productivity was also lost in profundal areas especially prior to World War II when open-water disposal was prevalent.

Losses of fish habitat were also substantial. Since 1926 approximately 6.2 km^2 of shallow littoral habitat along the south shore were obliterated by dredged material disposal and landfilling (Threader et al. 1986). Filling increased water depths along the Hamilton waterfront such that the only remaining littoral area is in Windermere Basin, and it is severely degraded. This, combined with the artifical nature of the shoreline, does not provide high quality fish habitat and limits fish production.

Populations of many warmwater fish including smallmouth bass, largemouth bass, rock bass, crappie, pumpkinseed, northern pike, bullhead, bowfin, and yellow perch, which are characteristic of highly productive shallow water marsh habitats, have declined from their historic maxima in the harbour (Whillans 1977, 1979; Holmes and Whillans 1984). Although there was some sport fishing pressure on largemouth bass, smallmouth bass, and northern pike, habitat degradation in the littoral zone is mostly responsible for these reductions. The kinds of degradation ranged from dredging and spoils disposal to thermal pollution, chronic toxicity, urban runoff, and nutrient enrichment, and the primary factors are therefore indistinguishable (Whillans 1979).

Other taxa such as whitefish, herring, and lake trout also inhabited the harbour historically (Whillans 1977, 1979; Holmes and Whillans 1984). Most were known to home to nearshore locations for spawning and/or feeding purposes. Sedimentation has occurred on the traditionally important spawning shoals along the north shore and on Randle's Reef along the south shore. Researchers from the Canada Department of Fisheries and Oceans have examined the north shore using an echolocator which penetrates the sediment and found that in some places, especially off LaSalle Park, there are echoes that may be rocky shoals buried under 30-60 cm of fine sediment (V. Cairns, Can. Dept. Fish. Oceans, Burlington, Ontario, pers. comm.). They were unable to find the shoal along the entire north shore, however, and there is some doubt (V. Cairns, pers. comm.) that it was a continuous shoal as indicated by Kerr and Kerr (1860-1898). Regardless, the loss of littoral habitat along the north and south shores presumably reduced the opportunities for spawning, feeding, and rearing, stressing coldwater fish populations in the harbour.

Turbidity is the most visible aspect of dredging and spoils disposal.

The available evidence suggests that turbidity will not create acute or chronic physiological problems for fish in the harbour, even at levels characteristic of intense dredging activity. Laboratory and field data indicate that most adult fish, larvae, and eggs can tolerate turbidity levels much higher (500-1000 mg/L; Sweeney 1978) than normally produced by dredging, although there is some evidence of reduced hatching success and growth rates in larval fish at these levels (Messieh et al. 1981).

The timing and location of dredging and disposal operations are important because turbidity plumes may seriously restrict seasonal fish movements. Historically, this may have been a contributing factor in the demise of the migrant coldwater fishery in the harbour. Large numbers of herring and whitefish immigrated into the harbour in November to spawn and in June to feed (Kerr and Kerr 1860-1898) and there is good evidence that lake trout emigrated to Lake Ontario in the fall to spawn (Whillans 1977). Maintenance dredging at or near the ship canal would have been very disruptive to these movements, temporarily restricting or completely eliminating access to key spawning and feeding habitat. Turbidity effects on fish movements within the harbour itself were probably mitigated by their restricted spatial and temporal nature.

Fish physically disturbed by noises or siltation at the excavation or disposal site, benthic fish in particular, will normally leave the site and return to recolonize it within a few hours after completion (Sweeney 1978). Some second order problems may arise due to changes in bottom configuration, depth, substrate, circulation, and sediment location (Acres 1983). These stresses may have had some influence on fish community transformations in the harbour, but they generally are not discernible from other kinds of habitat degradation, especially filling.

Siltation can eliminate all slow moving or sessile benthic organisms and spawning areas located within the excavation and disposal sites. Historically, the availability of unaffected habitat nearby probably had a mitigating influence on siltation effects until massive landfilling and spoils disposal began along the south shore. Currently the physical effects of dredging are probably insignificant to the fishery because the southeast region of the harbour is sparsely inhabited by carp and other species that are able to take advantage of badly disturbed habitats.

There is concern that dredging will remobilize contaminants from the sediments and enhance bioaccumulation and biomagnification in fish. Since the sulfide content of most of the harbour sediments is low, few metals will be present as insoluble sulfide compounds (OME 1978). Consequently, a reduction in dissolved oxygen and redox levels due to increased oxygen demand from resuspended sediments may lead to temporary increases in metal concentrations in the water column. The net effect on exposed fish would be an increase in the body burden of contaminant(s) as elevated uptake rates could only be partially compensated for by an increase in clearance rates (Acres 1983).

The acute and chronic effects of toxic chemicals released from the sediments by dredging are not clear. Many of the known contaminants, especially organochlorines such as PCBs, may affect the growth and photosynthesis of plankton (OME 1985). Other contaminants such as PAH's may act as mutagens and carcinogens in higher biota (Krahn et al. 1985). Contaminant analyses show that the intermediate size classes of white perch, carp, and brown bullhead in the harbour have low tissue concentrations of PCBs, mirex, mercury, and pesticides and are suitable for limited human consumption (OME and OMNR 1984). But the increased prevalence of tumours and some gill malformation reported in white suckers and brown bullheads from Hamilton Harbour (Great Lakes Water Quality Board 1983) are reasons for concern. In some species, for example white suckers, up to 40% of the specimens greater than 40 cm fork length have lip papillomas.

The effects of sediment associated contaminants on fish health are unknown. Laboratory toxicology studies indicate that many of the organic and inorganic chemicals present in Hamilton Harbour sediments can have acute and chronic adverse effects to fish and invertebrates. Little information exists describing the effects of combinations of chemicals at the individual, population, and community levels. Furthermore, many toxicological responses are confounded by compensating mechanisms within exposed populations. For example, reduced reproductive capacity may trigger density dependent compensatory mechanisms for increased fecundity. Fish populations may respond to contaminant stress by changes in growth, reproduction, behaviour, and mortality that are frequently indistinguishable from other habitat related stresses. There are several studies indicating that some fish species in Hamilton Harbour are responding to environmental stress (Great Lakes Water Quality Board 1983), but there are no data describing the adverse effects of stress on fish populations and communities in the harbour.

Although both dredging and disposal have been implicated in the degradation of the fish community, the persistent effects of spoils disposal in shallow littoral areas along the south shore is the more serious of the two. The loss of suitable littoral habitat placed the warmwater fish community under severe stress and caused several transformations (Whillans 1979) from which it has yet to recover. The coldwater offshore migratory species may have been stressed through the loss of spawning habitat and benthic productivity. The four indirect stresses (turbidity, oxygen depletion, nutrient enrichment, contaminant remobilization) may have had some local significance, but they were largely overshadowed by the direct disposal impacts. There is no evidence that present dredging and disposal practices in the harbour have any significant impact on the fisheries. This does not mean there are none, it simply means that they have not been detected. It seems likely that environmental effects of future dredging projects will also be negligible while the effects of spoils disposal may not if the current methodology is employed.

5. IMPACT ON FISHERIES REHABILITATION POTENTIAL

Future fisheries management in Hamilton Harbour presents a large challenge. The harbour has evolved from an area known for its coldwater and warmwater fisheries into a severely degraded system that is limited as a fish habitat (Whillans 1977, 1979). The most stressful perturbations during this transformation were deteriorating water quality and habitat destruction (Holmes and Whillans 1984). In spite of its current state there are indications that recovery is possible. Improving trends in water quality since the 1970s (Threader et al. 1986) suggest that solutions can also be found for the other environmental problems in the harbour. This has led to interest from fisheries agencies, particularly the Department of Fisheries and Oceans, in using the harbour as a demonstration project to illustrate the feasibility of fisheries rehabilitation in other degraded areas.

The development of a long range recovery program must address the three primary stresses impeding rehabilitation in the harbour: high nutrient loadings, highly contaminated sediments, and a lack of suitable fish habitat. Despite the recent improvements in water quality, the harbour remains one of the 42 areas of concern for priority cleanup in the Great Lakes. The highly contaminated sediments are largely responsible for this rating (Great Lakes Water Quality Board 1983). Other stresses including dredging and disposal, are secondary problems in that they influence sediment distribution and availability.

Holmes and Whillans (1984) and Holmes (1986) assessed the potential for fisheries rehabilitation in the harbour on the basis of available habitat and water quality. They concluded that management strategies combining measures to enhance and restore the system could be successful provided current improving trends in water quality continue. In only one area - the south shore between Pier 8 and Windermere Basin - did they suggest that rehabilitation was not feasible because there was no littoral habitat left and little likelihood that any could be provided. The rehabilitation potential of the other regions ranged from low (north and south of the ship canal) to excellent (northwest of Carroll's Point).

Although the extent of fisheries rehabilitation in Hamilton Harbour has not yet been decided, clearly the most pragmatic approach is to begin with the littoral areas frequented by the nearshore fish community (Holmes 1986). These highly productive areas are much reduced in quantity and quality from their historic maxima, but they are still utitized by warmwater fish species. In contrast, there is ample profundal habitat, however, its availability to coldwater species and its use are limited by oxygen depletion shortly after stratification. The dissolved oxygen problem has been under investigation since the mid-1970s (see OME 1985), but there is no apparent solution which would enable restoration of the historic coldwater fishery in the forseeable future (Holmes 1986).

The lack of suitable shoal habitat is also important to the potential rehabilitation of the coldwater fishery in the harbour. These shoals, along the north and south shores, were important spawning grounds for herring and whitefish populations in the harbour and western Lake Ontario (Kerr and Kerr 1860-1898). Restoration of the coldwater fishery in the harbour presumes that suitable shoals will be available. This is unlikely along the south shore because of current environmental conditions and the priority given to navigational uses. Further work is needed to determine the extent of the shoal along the north shore and the depth of burial. If rehabilitation of the coldwater fishery were to proceed, then reduced sediment loading, from Cootes Paradise in particular, will be required. Even then the abundance of coldwater fishes would be substantially lower than historic levels.

Current maintenance dredging operations are conducted to maintain navigable depths for shipping along the south shore. These activities per se have not had any documented impact on the fisheries in the harbour. Since future maintenance and capital dredging projects are expected to coincide with existing ship traffic areas, the effects on the fish community will likely be negligible and fisheries rehabilitation should not be affected to any great degree.

The greatest impact on rehabilitation potential lies in future spoils disposal practices. The present strategy of confined disposal began in the early 1970s because sediment contamination exceeded MOE guidelines for open-water disposal. This is expected to continue in the forseeable future since it is unlikely that sediment quality will improve appreciably. If filling continues at the present rate, then new containment areas will be required around the year 2000. This will be problematical because current construction techniques require shallow water depths. The only suitable areas for spoils disposal are along the east, west, and north shores of the harbour. These areas also have the highest fisheries rehabilitation potential. Fisheries habitat in Hamilton Harbour has been reduced to the extent that further destruction of littoral habitat could have far reaching repercussions on rehabilitation potential. There is little chance of fisheries restoration succeeding in the harbour unless the quantity and quality of existing nearshore areas are improved. Some filling is expected in Windermere Basin, but it will not seriously affect the fisheries or rehabilitation potential.

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